Bayer CropScience

EPA Correspondence No. 07-02A
May 8, 2007

Ms. Sherrie Kinard, Chemical Review Manager
Special Review and Reregistration Division (SRRD), 7504P
Office of Pesticide Programs
U.S. Environmental Protection Agency
Room S-4900, One Potomac Yard
2777 South Crystal Drive
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Re: Submission of a Report on the Effects Determination for the California Red-
    Legged Frog Potentially Exposed to Aldicarb in Response to the Stipulated
    Injunction Involving 66 Pesticides in California.

Dear Ms. Kinard:

Bayer CropScience
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In response to the Stipulated Injunction Involving 66 Pesticides in California
(71 FR 52073, September 1, 2006 and 72 FR 20544, April 25, 2007) Bayer
CropScience is submitting a report on the effects determination for the California
red-legged frog potentially exposed to aldicarb. Our submission is referenced as
follows:

MRID Number:___**47124601**_____

Fischer, D.L., T. Ramanarayanan, D. Moore, R. Thompson, and R. Breton. 2007.
Bayer CropScience. Research Triangle Park, NC. Report No. 201692. April 23,
2007. 138 pages.

Please phone me at 919-549-2870 or email me at larry.hodges@bayercropscience.com
if you have any questions regarding this submission.

Sincerely,

Larry R. Hodges, Ph.D.
Registration Manager, Insecticides

cc: Arty Williams
    Steven Bradbury

Bayer CropScience
Report 201692

<u>Study Title</u>
Effects Determination for California Red-Legged Frog
Potentially Exposed To Aldicarb

<u>Data Requirement</u>
None

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<u>Study Completion Date</u>
April 23, 2007

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<u>Bayer Report Number</u>
201692
Page *i* plus pages *ii* to *vii* for a total of 138 pages

STATEMENT OF NO DATA CONFIDENTIALITY

No claim of confidentiality is made for any information contained in this study on the basis of its falling within the scope of FIFRA 10(d)(1)(A), (B) or (C).

Company: Bayer CropScience

Company Agent: _L. R. Hodges_   Date __April 30, 2007__

Larry Hodges,
Regulatory Manager

These data are the property of Bayer CropScience, and as such, are considered to be confidential for all purposes other than compliance with FIFRA 10. Submission of these data in compliance with FIFRA does not constitute a waiver of any right to confidentiality, which may exist under any other statute or in any other country.

GOOD LABORATORY PRACTICE CERTIFICATION

Good laboratory practice requirements of 40 CFR Part 160 is not required for, and do not apply to, this document which is an ecological effects determination.

Submitter: Bayer CropScience

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Regulatory Manager

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Cantox Environmental, Inc.
Effects Determination for the California Red-Legged Frog Exposed to Aldicarb
Project Number: 88380

Page iii

EFFECTS DETERMINATION FOR CALIFORNIA RED-LEGGED FROG POTENTIALLY EXPOSED TO ALDICARB

Final Report

April, 2007

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Report 201692

Table of Contents

Appendices

List of Tables

List of Figures

1.0   Background

The California red-legged frog (*Rana aurora draytonii*) is endemic to California, and Baja California, Mexico. The species has been extirpated from 70 percent of its former range. Populations remain in approximately 256 streams or drainages in 28 counties in California. The Endangered Species Act (ESA) is the primary Federal law that provides protection for the California red-legged frog, given its listing as a threatened species in 1996.

The U.S. Environmental Protection Agency (EPA) must determine whether 66 pesticides currently authorized for use in California may adversely affect the California red-legged frog (CRLF). These effects determinations must be completed in three years in accordance with a recent settlement agreement. The purpose of this assessment is to make an "effects determination" for the federally listed California red-legged frog for direct and indirect effects associated with exposure to the insecticide aldicarb. The effects determination focuses on aldicarb-containing products that are produced by Bayer CropScience and that are registered for use in California.

2.0   Introduction

Aldicarb is one of the 66 pesticides under investigation. Aldicarb (2-methyl-2-(methylthio)priopionaldehyde O-(methylcarbamoyl)oxime; CAS No. 116-06-3; $C_7H_{14}N_2O_2S$) is a systemic insecticide sold only in granular form under the trade name TEMIK®. It is applied directly to the soil and is used to control mites, nematodes, and aphids on a variety of crops (e.g., cotton, potatoes, peanuts). It is registered for use on agricultural crops including citrus, cotton, dry beans, peanuts, pecans, potatoes, sorghum, soybeans, sugar beets, sugarcane, sweet potatoes, seed alfalfa, field grown ornamentals, and tobacco. Products containing aldicarb are not intended for sale to homeowners and there are no uses registered for residential areas.

Aldicarb was first registered for use in 1970. There is a perception of concerns surrounding residues of aldicarb in groundwater because of historical detects of aldicarb and its metabolites in drinking water wells. Initial concerns arose following residue detections in groundwater on Long Island leading to aldicarb being withdrawn from there in 1980. Since then there have been frequent refinements of aldicarb use patterns that have minimized aldicarb movement to groundwater. These include geographical prohibitions, soil restrictions, rate reductions and timing of applications as listed on the current labels. The most significant of these took place in 1989. These refinements have been supplemented by the adoption of a best management plan on the Florida ridge (Florida, 1994) and an active stewardship program (Hovis, 2005). Recent targeted monitoring programs for potable water in vulnerable areas have confirmed the effectiveness of these measures for protecting drinking water sources (Lenz *et al.*, 2006; Wyatt, 2006; Freeseman *et al.*, 2006; Mosier *et al.*, 2006; Lee *et al.*, 2006). Any detectable residues were low in concentration with none exceeding EPA drinking water standards (EPA 2004a).

Aldicarb is currently being reviewed by EPA for reregistration eligibility. Preliminary environmental fate and ecological effects (EFED) risk assessments for aldicarb reregistration eligibility were prepared by EPA in June 2001 (EPA, 2001) and January 2005 (EPA, 2005). Bayer provided EPA with a review of these EFED risk assessments on August 31, 2001 and May 6, 2005, respectively. EPA responded to some of the comments and concerns that Bayer raised in a revised EFED risk assessment that was made available for public comment on June 22, 2005 as part of Phase 3 of the aldicarb reregistration process (EPA, 2006a). Subsequent to commenting on the EFED risk assessments, Bayer provided EPA with higher tier risk assessments that built upon the screening-level methods used in the EPA assessments (CEI, 2006a,b). The higher tier assessments better characterized the risks posed by aldicarb to aquatic life and wildlife in areas where the pesticide is applied. EPA released a revised EFED risk assessment for aldicarb on September 27, 2006.

The purpose of this assessment is to make an "effects determination" for CRLFs exposed to formulated products containing aldicarb as produced by Bayer. The following assessment endpoints were evaluated: (1) direct toxic effects of aldicarb on the survival, reproduction, and growth of the CRLF; (2) indirect effects to CRLF prey resulting in reduced food supply; (3) indirect effects resulting from habitat modification (e.g., aquatic vascular plants), and (4) community structure of the plant community that constitutes aquatic breeding habitat, aquatic non-breeding habitat, upland habitat and dispersal habitat of the California red-legged frog. As part of the effects determination a conclusion of "no affect", "may affect, but not likely to adversely affect", or "likely to adversely affect" will be assigned to each of the assessment endpoints.

This effects determination was completed in accordance with guidance and methods described in the U.S. Fish and Wildlife Service (FWS) and National Marine Fisheries Service (NMFS) *Endangered Species Consultation Handbook* (FWS/NMFS, 1998), the August 5, 2004 Joint Counterpart Endangered Species Act Section 7 Consultation Regulations specified in 50 CFR Part 402 (FWS/NMFS, 2004; FR 69 47732-47762), the effects determinations for Barton Springs salamanders (EPA, 2006b), Alabama sturgeon (EPA, 2006c) and six Federally listed endangered species in Chesapeake Bay (EPA, 2006d), the Agency's Overview Document (EPA, 2004b), and the generic problem formulation document prepared for CropLife America (CEI, 2006c).

3.0   Problem Formulation

The objective of this problem formulation is to review the available information on aldicarb and the biology of California red-legged frog to identify the routes of exposure, assessment endpoints, measures of exposure and effect, and exposure scenarios that will be assessed. The problem formulation builds upon past assessments of aldicarb, including the EPA EFED assessment (EPA 2006a) and the refined aquatic life and wildlife risk assessments submitted to EPA by Bayer (CEI, 2006a,b). The problem formulation begins with the following sections:

> 3.1 – Use Patterns
> 3.2 – Physical and Chemical Properties

3.3 – Environmental Fate and Transport
3.4 – Species Profile of the California Red-Legged Frog
3.5 – Action Area
3.6 – Routes of Exposure
3.7 – Toxicity and Mode of Action
3.8 – Ecological Effects

The information from Sections 3.1 to 3.8 is used to create a generic conceptual model (Section 3.9), and identify assessment endpoints and measures of exposure and effects (Section 3.10 and 3.11). The information in Sections 3.1 to 3.8 is also used to develop the exposure scenarios (Section 3.12) that will be assessed in the analysis and risk characterization phases of the effects determination. The problem formulation concludes with an analysis plan (Section 3.13) outlining the approach that will be used to assess risks to the CRLF.

3.1 Use Patterns

In 2004, 4.53 million pounds of aldicarb active ingredient (a.i.) were applied to 5.45 million acres (Bayer CropScience, 2005). By weight, the greatest amount of aldicarb in the U.S. is applied to cotton (>50%), followed in order by peanuts, potatoes, sugar beets and citrus (EPA, 2006a). Aldicarb is registered for use on citrus, cotton, dry beans, grain sorghum, peanuts, pecans, potatoes, soybeans, sugar beets, sugarcane, sweet potatoes, field grown ornamentals, seed alfalfa, and tobacco, and is used in 28 states (EPA, 2006a). States with the highest aldicarb use are concentrated in the southeast and west, and include: Florida, North Carolina, South Carolina, Georgia, Washington, Idaho and California (EPA, 2006a). In Florida, aldicarb is applied predominantly to citrus groves. In the rest of the southern states, aldicarb is applied mostly to cotton and peanuts, while in the northwestern states it is applied mostly to potatoes, sugar beets and dry beans.

3.1.1 Application Rates and Methods

Aldicarb is applied only in granular form, and is formulated and sold as 10 and 15% a.i. granules. Acceptable application techniques always involve incorporation of the granules into the soil (EPA, 2006a). There is no lag between application and incorporation; they are done at the same time. Soil moisture is required to release the active ingredient from the granules. Application may occur at planting, post-emergent, or to established trees (EPA, 2006a). Application techniques at planting include: in-furrow, band, drill (just below seedline), and shank (EPA, 2006a). Side-dressing is the application method used when plants are emergent. Band application along the dripline is used for established trees (EPA, 2006a). Aldicarb is applied only once per year to most crops (the exceptions being cotton and sugar beets, which may receive both at-planting and post-emergence applications) (EPA, 2006a). Single application rates for aldicarb range from 0.15 lb a.i./A for cotton to 10.05 lb a.i./A for pecans (Bayer CropScience, 2005). Application methods result in <1-15% of granules being available on the soil surface. The most common methods of aldicarb application (in-furrow, drilled and shank methods) result in <1% of the granules being left at the soil surface (EPA, 2006a).

3.1.2 Aldicarb Use in California

The California Pesticide Use Database (CPUR) is one of the most extensive pesticide use databases available (see http://www.cdpr.ca.gov/docs/pur/purmain.htm) (Cal DPR, 2005). Since 1995, all agricultural pesticide use in California must be reported monthly to the county agricultural commissioner who, in turn, reports the data to the California Department of Pesticide Regulation (Cal DPR). These reports include the date and location (section, township, and range) where the application was made, the kind and amount of pesticides used and, if the pesticide is applied to a crop, the type of commodity. Identification numbers (IDs) for the site and the pesticide user ("operator") and the number of planted and treated acres (Cal DPR, 2005) are included. Before buying or using pesticides, every operator is required to obtain a unique operator ID from each county where pest control work will be performed. Growers obtain a site ID from the county agricultural commissioner for each location and crop/commodity where pest control work is anticipated, and it is recorded on the restricted material permit or other approved form. California has a broad definition of "agricultural use". Thus, reporting requirements include pesticide applications to parks, golf courses, cemeteries, rangeland, pastures, and along roadside and railroad rights-of-way. In addition, all post-harvest pesticide treatments of agricultural commodities must be reported, along with all pesticide treatments in poultry and fish production, and some livestock applications. Exceptions to the full use reporting requirements are home and garden use and most industrial and institutional uses (Cal DPR, 2005).

Data for aldicarb were downloaded from the CPUR database and imported into MS-Access 2003. Pesticide use data from 2005 were then queried to determine the amount of aldicarb (all products), as active ingredient and formulated product, used in each California County (Table 3-1). Counties reporting CRLF critical habitat or CRLF observations (April 1996 to May 2006) are also reported in the table.

Table 3-1. **Pesticide use and presence of CRLF critical habitat and CRLF observations (April 1996 to May 2006) by California county.**

County	CRLF Critical Habitat Present[1]	CRLF Observations 1996-2006[2]	Formulated Product Applied (lbs)[3]	Active Ingredient Applied (lbs)	Acres Treated
Fresno		Yes	441,913	66,287	66,372
Glenn			3,995	599	799
Imperial			22,951	3,443	3,252
Kern	Yes		205,494	30,824	26,878
Kings			510,432	76,565	66,575
Madera			27,027	4,054	3,498
Merced	Yes	Yes	186,010	27,902	23,229
Riverside		Yes	10,273	1,541	1,920
San Joaquin		Yes	400	60	80
Santa Clara	Yes	Yes	13	1	0.680
Sutter			4,500	675	820
Tulare			129,144	19,372	20,835

Source: Cal DPR, 2005; CNDDB, 2006; US DOI, 2006.
[1] - Indicates presence of CRLF Critical Habitat as defined by FWS April 2006 (US DOI, 2006).
[2] - Indicates observations of CRLFs from April 1996 to May 2006 (CNDDB, 2006).
[3] - Includes both aldicarb and aldicarb (or other similar) categories in the PUR database.

Bayer CropScience
Report 201692

Table 3-2 presents the distribution of aldicarb use in California for agricultural and non-agricultural purposes in 2005. In California, the use of aldicarb on cotton accounts for 99% of the use, with 230,399 lbs of active ingredient (a.i.) applied and 213,611 acres treated in 2005. There is a sharp drop in use to the next crop pecans with 456 lbs of active ingredient applied and 96 acres treated. The remaining uses of aldicarb in California reported for 2005 were on beans, sorghum, ornamental flowers, and sugar beets.

Table 3-2.   Crop-specific aldicarb use in California in 2005.

Crop	Formulated Product Applied (lbs)	Active Ingredient Applied (lbs)	Acres Treated
Cotton, general	1,535,993	230,399	213,611
Pecans	3,042	456	96
Soil application, Preplant-Outdoor (Seedbeds, etc.)	1,332	200	222
Beans, dried-type	1,327	199	296
Sorghum/Milo general	440	66	32
Non-greenhouse grown cut flowers or greens	12	1	0.549
Sugar beet, general	5	1	3
Non-outdoor grown cut flowers or greens	2	0	0.131

Source: Cal DPR, 2005; CNDDB, 2006; US DOI, 2006.
NR - Not reported in California Pesticide Use Report Database

Figure 3-1 shows the trends in aldicarb use in California from 1995 to 2005 on the basis of pounds used and acres treated. Alidcarb use was highest from 1995 to 1998 and declined from 1998 to 1999. Since 1999, the amount of aldicarb used and number of acres treated has remained fairly consistent.
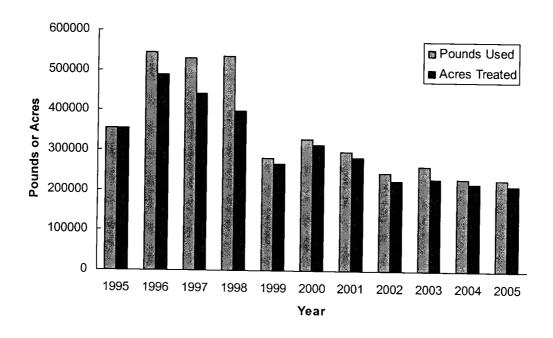
Figure 3-1. Pounds of aldicarb active ingredient used and acres treated in California for agricultural and non-agricultural uses from 1995 to 2005 (Cal DPR, 2005).

Aldicarb is sold under the trade name TEMIK® and is only produced in a granular form containing 15% active ingredient. TEMIK® products containing 10% active ingredient are no longer produced or sold by Bayer. TEMIK® products that are permitted for use in California are presented in Table 3-3 and include TEMIK® 15G, TEMIK® 15G CP, and TEMIK® 15G (for sale in CA only).

Table 3-3.   Currently registered aldicarb end-use products.

Registration No.	Registration Name	% Active Ingredient
264-330	TEMIK® Brand 15G Aldicarb Pesticide	15
264-417	TEMIK® Brand 15G CP Aldicarb Pesticide	15
264-426	TEMIK® Brand 15G Aldicarb Pesticide (CA Only)	15

Aldicarb is a restricted use pesticide with acute toxicity and groundwater contamination concerns. The pesticide labels for TEMIK® products provide guidance for minimizing potential off-site transport following application. The labels report the states and counties where use of TEMIK® is not permitted. For example, TEMIK® is not to be used in the Californian counties of Del Norte or Humboldt, and it is not to be applied to citrus in California. In California, TEMIK® can only be applied between March 1 and September 1, and it must not be applied within 50 feet of any drinking water well. Other application restrictions specific to California are reported on the respective labels. Application rates for the uses of TEMIK® that are permitted in California are presented in Table 3-4. Application rates range from 0.53 lbs a.i./acre for cotton to 4.95 lbs a.i./acre for pecans. The application of TEMIK to pecans is permitted under a special local need registration (No. CA-990026). The use of TEMIK® on citrus, potatoes and sugarcane is not permitted in California, but is allowed in other States.

Table 3-4.   Crop uses and application rates for TEMIK®.

Crop/Time of Application	Range of Application Rates (lbs a.i./Acre)	Application Method	Application Instructions
Cotton At planting	0.53 to 1.05	Drilled or in-furrow	Do not exceed total of 3.15 lbs a.i./acre/season in CA.
At first squaring	1.05 to 2.1	Side-dress	
From squaring through early bloom	2.1	Side-dress	
Dry beans At planting	0.75 to 1.05	In-furrow or band with immediate soil-incorporation	
Peanuts At planting	1.05 to 3.0	In-furrow or band and work into soil	In southwest, use high rate only.
Pecans			

Table 3-4.   Crop uses and application rates for TEMIK®.

Crop/Time of Application	Range of Application Rates (lbs a.i./Acre)	Application Method	Application Instructions
Bud break to nut set	2.55 to 4.95	In-furrow or band and work into soil	Special local need registration for TEMIK® 15G Lock'n Load.
Sorghum At planting	1.05	In-furrow	
Soybeans At planting	0.75 to 3.0	Drilled or in-furrow	
Sugar beets At-planting	1.05 to 2.1	Band and work into soil or drilled	
At-planting and post-emergence (split application)	2.1 + 2.1	Band, drilled, or side-dressed	
Post-emergence	1.05 to 2.1	Band or side-dressed	Do not make more than one 'at-planting' application and two post-emergence applications per crop. Do not exceed a total of 4.2 lbs a.i./acre.
Sweet potatoes At planting	1.5	Band and cover immediately	Do not exceed a total of 3.0 lbs a.i./acre.

3.2   Physical and Chemical Properties

Aldicarb is a white crystalline solid and is part of the carbamate ester family. Under normal conditions the active ingredient is heat sensitive and inherently unstable. The physical and chemical properties of aldicarb are presented in Table 3-5. Aldicarb is sold under the trade name TEMIK® and is only produced in granular forms containing 15% active ingredient (see Table 3-3). TEMIK® granules are primarily formulated with a gypsum core; corn cob grit is used to a much lesser extent. Aldicarb is attached to the core using a vinyl binder. Post-formulation, a black coating of graphite or carbon is applied over the granule.

Table 3-5.   Physical and chemical properties of aldicarb[a].

Physical – Chemical Property	Aldicarb
Chemical Name	2-methyl-2-(methylthio)priopionaldehyde O-(methylcarbamoyl)oxime
Common Name	Aldicarb
CAS No.	116-06-3
Molecular Formula	$C_7H_{14}N_2O_2S$
Molecular Weight	190.3
Density	1.2 g/cm^3
Melting Point	100°C
Boiling Point	Decomposes above 100°C
Vapor Pressure	2.55 x 10^{-5} mg Hg @ 25°C
Water Solubility	6,000 mg/L (pH 7, 25°C)

Table 3-5.   Physical and chemical properties of aldicarb[a].

Physical – Chemical Property	Aldicarb
Henry's Law Constant	1.7×10^{-10} atm m^3/mole
Bioconcentration factor (BCF)	1.62[b]
Octanol-Water Partition Coefficient (K_{ow})	13.5[c]
Carbon Matter Partition Coefficient (K_{oc})	100

[a] EPA (2006a).
[b] Estimated value using equation in Arnot and Gobas (2004). See Section 4.1.2.2.
[c] Hansch and Leo 1985; Howard 1991.

Aldicarb is released rapidly from TEMIK® granules upon contact with water and a near disintegration of the granules can be expected within one week of application (Krebs, 2001). Based on the properties presented in Table 3-5, aldicarb is considered highly water soluble (6,000 mg/L) and mobile in soils (K_{oc} = 100). The octanol-water partition coefficient (K_{ow} = 14) suggests that aldicarb has a low potential for partitioning to organism lipids. The Henry's Law Constant value indicates that volatilization from surface waters is not an important fate process.

Aldicarb has 18 known metabolites and degradation products, only two of which are toxic to aquatic and terrestrial animals: aldicarb sulfoxide and aldicarb sulfone (the carbamate metabolites). The carbamate metabolites normally degrade to the non-carbamate, non-toxic aldicarb oxime and aldicarb nitrile. These degradation pathways are the same in plants, animals and soil (Jones and Estes, 1995). The EFED assessment noted that valid studies with data reporting decline rates of aldicarb sulfoxide and sulfone are currently lacking. However, in an aerobic soil metabolism study using sandy loam soil the half-life of aldicarb was 2.3 days. The major transformation products were aldicarb sulfoxide and aldicarb sulfone accounting for 85 to 90% of the applied pesticide (EPA, 2006a; MRID 44005001). An oral dose of 50 µg of TEMIK® administered to rats was metabolized and excreted through the urine as aldicarb sulfoxide (~ 40%), oxime sulfoxide (~ 30%), and 5 to 9 polar compounds (~ 30%) (Risher et al., 1987). Based on the rapid transformation of aldicarb to these metabolites and their increased persistence in the environment compared to parent compound (Bull et al., 1970; Smelt et al., 1979), the toxic metabolites aldicarb sulfoxide and aldicarb sulfone were also considered in the effects determination.

3.3   Environmental Fate and Transport

3.3.1   Soil

Aldicarb is soluble, mobile and moderately persistent in soil (Kidd and James, 1991; HSDB, 2003). Aerobic soil metabolism is the primary dissipation route for parent aldicarb in unsaturated soil (MRID 44005001). Reported half-lives for parent aldicarb in aerobic soil range from 1 to 28 days (MRIDs 00102051, 00093642, 00080820, 00093640, 00053366, 00101934, 00035365, 00102071, 45739801, 45739802). Laboratory studies indicate that parent aldicarb and the carbamate metabolites degrade to oximes and nitriles in aerobic soils with a half-life of up to three months. The processes behind this degradation depend on soil conditions, but most often involve soil-catalyzed hydrolysis and microbial decay (Cohen, 1986; Lightfoot et al., 1987; Bank and Tyrrell, 1984). The

ability to metabolize aldicarb and the sulfoxide is common in soil fungi (Jones, 1976). Soil temperature is the dominant variable influencing aldicarb degradation under normal agricultural conditions (Jones, 1976). Field degradation studies indicate that aldicarb and the carbamate metabolites have field half-lives between 0.3 to 3.5 months (Jones and Estes, 1995).

Soil moisture is a critical factor in aldicarb stability. Formulated aldicarb is relatively stable in dry soil. Fifty-percent moisture is optimal for the oxidation of aldicarb, and 100% moisture (i.e., soil saturation) causes rapid decomposition to non-toxic products (Bull et al., 1970). Studies have shown that aldicarb degradation rates are comparable under anaerobic and aerobic soil conditions. However, degradation of the sulfoxide and sulfone has been shown to be more rapid (5.1 to 131 d) in anaerobic soil (Sheets and Hirsh, 1976; Smelt et al., 1983). Lightfoot et al. (1987) conducted a laboratory study investigating the degradation of aldicarb and the carbamate metabolites in water and soil (Table 3-6). Their results indicated that the metabolites are more persistent under all conditions. Carbamates were most persistent in sterilized water (pH was within one unit of neutral throughout the experiment for all water treatments), and least persistent in the plough-layer soil. Studies of photolysis on soil surfaces have not been conducted because aldicarb is incorporated into the soil (FAO/WHO, 1994).

Table 3-6.   Degradation rates for aldicarb, aldicarb sulfoxide, and aldicarb sulfone (Source: Lightfoot et al., 1987).

Matrix	Aldicarb Half-life at 25°C (d)	Total Carbamates Half-life at 25°C (d)
Plough-layer soil		
sterilized	2.5	10
unsterilized	1.0	24
Soil water		
sterilized	1679	1924
unsterilized	156	175
Distilled water (no buffer)	671	697
Saturated zone (soil and water)		
sterilized	15	16
unsterilized	37	123

Aldicarb and the carbamate metabolites are generally mobile in soil (Cohen, 1986), though to varying extents. Mobility depends on soil characteristics, pH, temperature and extent of leaching, rainfall patterns, and other factors (Coppedge et al., 1977; Bowman, 1988). Based on Freundlich K values, parent aldicarb is the most mobile, followed by sulfoxide and sulfone (EPA, 2006a). Leaching is most extensive in soils with low organic content (Cohen, 1986).

Studies have indicated that volatilization of aldicarb and its degradation products from soil, although minor, is influenced by soil moisture. As the rate of water evaporation increases so do the volatilization rates of aldicarb and the degradation products (Bull et al., 1970; Coppedge et al., 1967; Richey, 1972; Supak, 1972; Supak et al., 1977). Temperature has also been shown to be positively correlated with volatilization of aldicarb and the degradation products from dry and wet sand (Bull et al., 1970).

However, volatilization from soil is limited and minimized by incorporation of the granules in soil. The potential for air contamination is negligible (Supak *et al.*, 1977).

3.3.2 Water

Aldicarb is unlikely to enter the aquatic environment in granular form because: 1) aldicarb is released rapidly from the granules upon contact with soil moisture and the granules disintegrate within a week, and 2) it is applied to fields, not to water bodies. Incorporation of granules into the soil limits runoff losses. Because aldicarb and the carbamate metabolites do not significantly bind with inorganic soil particles, residues tend to move with soil water (Jones and Estes, 1995). Organic material in soil tends to inhibit this process by binding with aldicarb residues. Movement in the unsaturated zone is usually downward. If dissolved aldicarb residues reach the saturated zone they may move in a horizontal direction, with the interflow of the water, at a rate equivalent to the water's velocity (generally between 0.03 and 0.5 m per day) (Jones and Estes, 1995). Residues normally degrade before reaching aquifers, due to rapid degradation. In cases where aldicarb is transported via interflow out of the soil matrix, it generally seeps into adjacent ditches, or is constrained to shallow groundwater near the treated site (Jones and Estes, 1995). In surface waters, aldicarb is not volatile (Lyman *et al.*, 1990) and is not expected to bind with sediments (HSDB, 2003).

Aldicarb is susceptible to hydrolysis, a property consistent with it being a carbamic acid ester (FAO/WHO, 1994). The hydrolysis rate for aldicarb depends on pH (Given and Dierberg, 1985). Reaction rate constants for aldicarb residues have been shown to remain relatively constant below pH 7, and to increase rapidly as pH rises above neutral, resulting in half-lives of less than two years in most environments (Hansen and Spiegel, 1983). Studies indicate that parent aldicarb is quite stable in neutral and acidic water (half-lives between 240 and 445 d at 15°C for pHs between 7.5 and 5.5, respectively). Aldicarb hydrolyzes more rapidly at pH 8.5 (7 d) and, in slightly alkaline conditions (pH 8 and 9), aldicarb carbamates are rapidly degraded to their relatively non-toxic oximes and nitriles (Andrawes 1976a,b,c; Tobler, 1970; Hansen and Spiegel, 1983). Aldicarb sulfoxide and sulfone are generally more susceptible to hydrolysis at higher pHs. Half-lives of 2,477 and 821 days were reported for aldicarb sulfoxide and sulfone, respectively, in distilled water (23°C) with a pH of 6. At the same temperature and a pH of 9.0, the half-lives decreased to 2.2 and 0.9 days, respectively (Lightfoot *et al.*, 1987).

Field studies indicate that it is unlikely for any significant aldicarb sulfoxide or aldicarb sulfone to occur in open bodies of water, and thus photodegradation in water has not been considered for these metabolites (Bayer CropScience, 2005). Aqueous photolysis degrades parent aldicarb to oxime and nitrile ($t_{1/2} = 4$ d) in clear, shallow water (MRID 42498201). In pond water, aldicarb has a half-life of between 5 and 10 days (Baron, 1991). The half-lives of aldicarb, aldicarb sulfoxide and sulfone in aerobic aquatic environments have been found to be 5.5, 5 and 3.5 days, respectively (MRID No. 45592107). In anaerobic aquatic environments, degradation can be slightly faster for the carbamate metabolites, with half-lives of 3.4 and 3.85 days for the sulfoxide and sulfone, respectively. The half-life of aldicarb residues in water can range from days to years depending on the environmental conditions (e.g., redox potential, presence of microbes,

light, pH and temperature) (Howard, 1991). However, monitoring studies indicate that, in most areas, aldicarb residues degrade in the upper portion of the unsaturated zone in soil, before ever reaching aquatic environments (Jones and Estes, 1995).

3.3.3 Plants

In plants, aldicarb is metabolized by oxidative processes that degrade the parent molecule to the sulfoxide and sulfone. In addition, hydrolysis degrades the carbamates to oximes, and, ultimately, to the nitrile (IPCS, 1991). Biotransformation of aldicarb to sulfoxide in plants has been shown to occur over a matter of days (e.g., Metcalf *et al.*, 1966; Bull, 1968). Oxidation from the sulfoxide to the sulfone occurs more slowly (Coppedge *et al.*, 1967). The physical-chemical properties of aldicarb, aldicarb sulfoxide and aldicarb sulfone indicate that these compounds are unlikely to occur in aquatic plants and algae at concentrations that exceed those in water. The low K_{ow} (13.5 ml/g) and rapid hydrolysis rates (T ½ = 3.5 to 5.5 days) of these compounds will limit there presence and persistence in surface waters. Aldicarb and the toxic metabolites are considered to have a low bioconcentration potential and do not accumulate in tissues over long-term exposures (Howard, 1991; Smith, 1992; Johnson and Finley, 1980; WHO, 1991; EPA, 2006a). Thus, accumulation in aquatic plants and algae are not expected.

Andrawes *et al.* (1971) investigated the metabolism of radio-tagged aldicarb in field-grown potatoes after an in-furrow application. Thirty days after treatment, the highest level of total aldicarb residues was found in the seed, but after 90 days, the highest level was found in the foliage. In a similar study, Andrawes and Bagley (1970) investigated the metabolism of aldicarb in greenhouse-grown sugar beet plants. Ninety days following the original application, no parent aldicarb was present in the plant tissue. Approximately 40% and 20% of applied radioactivity was found in the foliage and roots, respectively, in the form of carbamate metabolites. At 140 days following the initial application, the amount of sulfone present had increased relative to sulfoxide, particularly in the foliage. Sun *et al.* (2004) conducted a laboratory study investigating plant uptake and subsequent loss of aldicarb and metabolites in corn, mung bean and cow pea. Results indicated that peak plant tissue concentrations occurred 2 to 10 days after application. These results are consistent with the findings of Andrawes *et al.* (1973) who investigated the fate of aldicarb in field-grown cotton. The highest amounts of aldicarb and aldicarb sulfoxide were recorded on the first measurement, nine days after treatment. Aldicarb sulfone reached a peak 22 days after the application. The parent aldicarb was no longer detectable at day 58, and the carbamate metabolites were at low concentrations by day 146. Similar results have been found in other crops (e.g., tobacco, Khasawinah and Hirsh, 1976; peanuts, Andrawes, 1972).

3.3.4 Animals

A number of studies on mammalian species have shown that aldicarb and the carbamate metabolites are readily, and almost completely absorbed from the gastrointestinal tract (Knaak *et al.*, 1966a,b; Andrawes *et al.*, 1967; Dorough and Ivie, 1968; Dorough *et al.*, 1970; Hicks *et al.*, 1972; Cambon *et al.*, 1979). Studies also indicate that mammals quickly excrete aldicarb (Baron, 1991).

In a study with female rats orally administered 0.4 mg/kg body weight of aldicarb, 94.1% of the dose was excreted within 5 days (87.5% in urine and 6.6% in faeces). On the first day following dosing, the dominant metabolite in the excreta was aldicarb sulfoxide. By day 3, and through to day 7, the dominant metabolite in excreta was nitrile sulfone. Only a small trace of parent aldicarb was found in the excreta, suggesting rapid degradation of the compound (Andrawes *et al.*, 1967). In a comparable study, female beagle dogs were dosed for 20 days with 0.75 mg/kg bw/day of aldicarb. Elimination via the urine averaged 74% of the dose, and was nearly complete within 11 days (Sullivan and Carpenter, 1968). Metabolic products were the same as in rats.

Lactating cows dosed for 14 days with 0.12 to 1.2 mg/kg aldicarb in their diet excreted more than 90% of the radio-tagged aldicarb residues in their urine. Two to three percent was excreted in faeces, and less than 2% was eliminated during milking. The relative amounts of various metabolites were similar throughout the study, and there was no significant variation with dose (Dorough *et al.*, 1970). A similar study was conducted in goats dosed with 0.165 mg/kg body weight per day for 10 days. Milk production and quantity of excreta remained stable throughout the study. On day two of the study, the dominant metabolite in the urine was aldicarb oxime sulfoxide. Similar to other mammals, by day 5 and through to day 10, aldicarb nitrile sulfone was the dominant metabolite in urine (Andrawes and Lee, 1986).

Hicks *et al.* (1972) treated 10 hens with a 1:1 molar mixture of aldicarb and aldicarb sulfone at a concentration of 0.66 mg/kg body weight. Over 85% of radio-tagged aldicarb was excreted within 10 days. Only a small amount remained in eggs and tissue (<0.05%). Analysis of metabolites in the faeces indicated that the metabolic pathway in chickens is similar to that in mammals (i.e., no aldicarb was present, but sulfoxide, sulfone, oximes and other metabolites were present).

3.4 Species Profile of the California Red-Legged Frog

3.4.1 Species Listing Status

The U.S. Fish and Wildlife Service (FWS) listed the California red-legged frog (CRLF) (*Rana aurora draytonii*) as a threatened species on June 24, 1996. This rule does not extend to CRLFs that inhabit:

1. The state of Nevada.
2. Humboldt, Trinity, and Mendocino counties, California.
3. Glenn, Lake, and Sonoma counties, California, west of the Central Valley Hydrological Basin.
4. Sonoma and Marin counties north and west of the Napa River, Sonoma Creek, and Petaluma River drainages, and north of the Walker Creek drainage.

The FWS has given the California red-legged frog a recovery priority number of 6C. This code identifies the species as having a high degree of threat and a low potential for recovery. Threats to the CRLF include, but are not limited to trematode and chytrid fungal disease, direct and indirect impacts from some human recreational activities, flood

control maintenance activities, water diversions, unmanaged overgrazing activities, competition and predation by nonnative species (e.g., warm water fish, bullfrog), habitat removal and alteration by urbanization, and some agricultural pesticides and fertilizers (FWS, 2006). All of these stressors contribute to the existing Environmental Baseline for California red-legged frog.

3.4.2   Description and Taxonomy

The California red-legged frog is endemic to California and Baja California, Mexico. It is one of two subspecies of red-legged frog (*Rana aurora*). The other is the northern red-legged frog (*R. a. aurora*) that ranges from Vancouver Island, British Columbia, south along the Pacific coast to northern California (FWS, 2002a). The CRLF is the largest native frog in the western United States (Wright and Wright, 1949).

3.4.3   Distribution

The historical distribution of the California red-legged frog is believed to have included 46 counties in California from the Point Reyes National Seashore, Marin County, California, and inland from Redding and Shasta County, California, south to northwestern Baja California, Mexico (FWS, 2002a, 2006). The CRLF has been extirpated from 24 of these counties accounting for 70% of its former range (FWS, 2002a, 2006). The current distribution of the CRLF includes the coastal drainages of central California, from Marin County, CA, south to northern Baja California, Mexico, and in a limited number of drainages in the Sierra Nevada, northern Coast, and northern Transverse Ranges (Figure 3-2) (FWS, 1996, 2002b, 2006).

Figure 3-2.   Current distribution of the California red-legged frog by county (FWS, 2002a).

The FWS recovery plan summarizes the present status of the California red-legged frog in different portions of its current range (FWS, 2002a). This information is useful in understanding the current Environmental Baseline for CRLF.

Sierra Nevada Foothills and Central Valley
Most of this region has not been surveyed, thus the true status of the CRLF is unknown. CRLFs have been observed in a few drainages in the foothills of the Sierra Nevada. In Butte County, CRLF populations have been documented in French and Indian Creeks. These populations are on private lands near the Plumas National Forest (FWS, 2002a). In 2000, another population of CRLFs was discovered in this county on the Feather River Ranger District of the Plumas National Forest (FWS, 2002a). Populations of CRLFs have also been reported in El Dorado County (1997 and 1998), and in 2001 a single CRLF was observed in Placer County on U.S. Forest Service land near the confluence of the Rubican River and middle fork of the American River (FWS, 2002a).

North Coast Range Foothills and Western Sacramento River Valley
CRLF have historically been observed in the tributaries of several counties in this recovery unit, including Glenn Colusa, and Lake Counties (FWS, 2002a). More recently, sightings have been reported in upper and lower Napa and Lake Counties.

North Coast and North San Francisco Bay
Populations of CRLFs occur around Point Reyes in Marin County, including locations in Point Reyes National Seashore and the Golden Gate National Recreation Area (FWS, 2002a). CRLFs have also been observed on Mount Tamalpais and the Tiburon Peninsula in Marin County. A large breeding population of CRLFs occurs in Ledson Marsh in Annadel State Park, Sonoma County. Three occurrences have been reported in Solano County near Suisun Marsh (FWS, 2002a).

South and East San Francisco Bay
The most recent sighting of CRLF in San Francisco County occurred in 1993, in Golden Gate Park. These populations face severe barriers that are expected to inhibit dispersal between populations (FWS, 2002a). Populations are known to occur in the canals near the San Francisco International Airport in San Mateo County. CRLF reproduction has been confirmed for some of the populations.

Contra Costa and Alameda Counties contain most of the known CRLF populations in the San Francisco Bay area. Healthy populations of CRLFs occur in the eastern portions of Contra Costa and Alameda Counties (FWS, 2002a). Many of the ponds and creeks found in the Simas Valley in Contra Costa County support populations of CRLF (FWS, 2002a). Recent CRLF sightings have been made in ponds and seeps in the foothills of Mount Diablo, Contra Costa County. Populations have also been observed in Corral Hollow Creek in San Joaquin County and near the San Joaquin/Alameda County border (FWS, 2002a).

Central Coast
The Central Coast region spans San Francisco to Santa Barbara County and has the greatest number of drainages currently populated by CRLF (FWS, 2002a). Most of the coastal drainages of San Mateo and Santa Cruz Counties support populations of CRLF. CRLFs are found throughout Monterey County in nearly every coastal drainage system. In San Luis Obispo County, CRLFs are found in suitable water bodies on the coastal plain and western slopes of the Santa Lucia Range (FWS, 2002a).

Diablo Range and Salinas Valley
The CRLF was once abundant in the inner Coast ranges between the Salinas River system and the San Joaquin Valley (FWS, 2002a). It currently occupies ≤10% of its historic range in these localities. Several populations of CRLF occur on the eastern side of the Diablo range in creeks in Fresno and Merced Counties (FWS, 2002a). In Monterey County, CRLF occur in the Elkhorn Slough watershed.

Northern Transverse Ranges and Tehachapi Mountains

This region is comprised of all of Santa Barbara and parts of Ventura, Los Angeles and Kern Counties. CRLFs occur on the Santa Maria River, Santa Barbara County, up and downstream of the Twitchell Reservoir (FWS, 2002a). Locations to the south (San Antonio Creek, Terrace, and Lagoon) are considered among the most productive CRLF locations in Santa Barbara County (FWS, 2002a). Most of these locations are found on Vandenberg Air Force Base. The habitat in this area has been relatively undisturbed and there are few occurrences of exotic species (e.g., bullfrogs). The largest populations in the northern Transverse Range are located on creeks that flow into the Cuyama and Sisquoc Rivers (FWS, 2002a). Poor habitat and introduction of aquatic predators have resulted in smaller populations of CRLFs in the Santa Ynez River Basin in Santa Barbara County. Recent surveys for CRLFs in the Tehachapi Mountains are not available (FWS, 2002a).

Southern Transverse and Peninsular Ranges

The California red-legged frog is native to parts of Los Angeles, San Bernardino, Orange, Riverside, and San Diego counties (FWS, 2002a). In 1999, a population of 15 to 25 adults was reported in the Angeles National Forest, Los Angeles County. Non-native predators, disease and parasites threaten this population (FWS, 2002a). A breeding population of 20 to 25 adults, 10 to 15 juveniles and several hundred tadpoles was recently discovered in East Las Virgenes Creek, Ventura County. South of the Tehachapi Mountains, CRLFs are currently known to occur in Amargosa Creek, Los Angeles County, and Cole Creek, Riverside County (FWS, 2002a). Bullfrog predation is believed to be the reason for the reduction in population size.

3.4.4 Habitat

California red-legged frogs use a variety of aquatic, riparian, and upland habitats from sea level to an elevation of 1,500 meters (FWS, 2002b). Dispersal and habitat use depend on climate, habitat suitability, and life stage (FWS, 2002a). Preferred breeding and summer habitat includes still or slow-moving permanent streams with deep water (>0.7 meters) and dense riparian vegetation (FWS, 2002a, 1996). Alternate habitats include marshes, ponds, damp woods and meadows. California red-legged frogs will breed in artificial impoundments such as stock ponds (FWS, 2002b). The CRLF is active year-round in coastal areas (Bulger et al., 2003). Upland summer habitats include small mammal burrows and moist leaf litter (Jennings and Hayes, 1994), the underside of boulders, rocks, and debris, various agricultural features (FWS, 2002a), and cracks in the bottom of dried ponds (FWS, 2002a).

During the summer, some CRLFs may leave breeding areas and migrate to upland habitats. Research has focused on CRLFs in aquatic habitats and little is known about their terrestrial movements. Bulger et al. (2003) studied the terrestrial movements of CRLFs inhabiting a coastal watershed in Santa Cruz County, California. This study examined the use of terrestrial habitats in relation to season, breeding chronology, and precipitation. Over 75% of the individuals monitored traveled short distances to upland areas following rain events, but returned to aquatic habitat after a short period (Bulger et al., 2003). Ninety percent of these individuals remained within 60 meters of water at all

times (Bulger *et al.*, 2003). The authors referred to these individuals as non-migrating frogs. Non-migrating frogs were almost always within 5 meters of their summer aquatic habitat, but would move as far as 130 meters upland during rain events for a median period of approximately 4 to 6 days (Bulger *et al.*, 2003). The higher levels of rain that occur in November and early December increase the median distance of CRLFs from water (15 to 25 meters) and median time in upland habitats (20 to 30 days). CRLFs make little use of upland habitats as winter passes and the breeding season approaches (mid December) (Bulger *et al.*, 2003). From February to May, 90% of the non-migrating frogs remained within 6 meters of water (Bulger *et al.*, 2003).

The remainder of the adult population (<25%) made additional overland trips between different aquatic sites and were referred to as migrating frogs. Twenty-five migration events, ranging from 200 to 2,800 meters, were observed (Bulger *et al.*, 2003). CRLFs traveled shorter distances (<300 meters) in 1 to 3 days and took up to 2 months to complete longer journeys (Bulger *et al.*, 2003). These migrations occurred through coniferous forests and agricultural and range lands (Bulger *et al.*, 2003). Rather than using corridors, CRLFs followed straight-line migrations between habitats (Bulger *et al.*, 2003). The authors estimated that 11 to 22% of the adult population made annual migrations from their breeding habitat. The study suggested that adequate protection of CRLFs could be accomplished by maintaining suitable habitat within 100 meters of aquatic sites and managing human activities on a seasonal basis in these areas (Bulger *et al.*, 2003).

3.4.5 Life History and Ecology

The following sections describe the physical characteristics, foraging behavior, and reproduction of the California red-legged frog.

3.4.5.1 Body Size

The California red-legged frog is the largest native frog in the western United States (Wright and Wright, 1949). Adult females are generally longer than males (F: 8.7 to 13.8 cm, M: 7.8 to 11.6 cm) (Hayes and Miyamoto, 1984). Larvae range in length from 1.4 to 8.0 cm (Storer, 1925). Bulger et al. (2003) reported body weights for male and female California red-legged frogs ranging from 48 to 214 g. In a ten year study in San Luis Obispo County, California, Scott and Rathbun (2001) collected body length and weight data for 459 California red-legged frogs. Body lengths ranged from 3.5 to 13.9 cm and weights ranged from 4.3 to 247 g. USGS (2004) conducted a survey of CRLF in Big Lagoon, Golden Gate National Recreation Area from 2002 to 2003. Big Lagoon is a wetland project area located in the Point Reyes Critical Habitat Unit #12. Nine male CRLFs were caught during the study. Their body lengths ranged from 8.2 to 9.5 cm with mean and median length of 8.7 cm. Body weights for the nine male CRLFs ranged from 54.7 to 94.0 g with mean and geometric mean body weights of 76.5 and 75.3 g, respectively (USGS, 2004).

3.4.5.2 Diet

The foraging behavior of the CRLF is highly variable and is defined by life stage and habitat (Hayes and Tennant, 1985; FWS, 2002a). The diet of larvae has not been well

studied, but they are primarily algal grazers (FWS, 2002a). They also consume organic debris, plant tissue and minute organisms (NatureServe, 2006). Their anatomy enables them to filter and entrap suspended algae (Seale and Beckvar, 1980) and their mouthparts are designed for effective grazing of periphyton (Wassersug, 1984; Kupferberg et al., 1994; Kupferberg, 1997; Altig and McDiarmid, 1999). Some of the more common food items consumed by larvae include filamentous green algae (Dickman, 1968), filamentous blue-green algae (Pryor, 2003), epiphytic diatoms (Kupferberg, 1997) and detritus and various other algae (Jenssen, 1967). Larvae are also known to feed on algal species that are considered nuisance species or form blooms (Bold and Wynne, 1985).

Adult CRLFs consume a variety of invertebrate and vertebrate species found along the shoreline and on the water surface. They will also forage several meters into dense riparian vegetation along the shoreline (FWS, 2002a). A study examining the gut contents of 35 CRLFs reported prey from forty-two taxa (Hayes and Tennant, 1985). The prey groups observed most often included carabid and tenebrionid beetles, water striders, lycosid spiders, and larval neuropterans (Hayes and Tennant, 1985). The most commonly observed prey species were larval alderflies (*Sialis* cf. *californica*), pillbugs (*Armadillidrium vulgare*), and water striders (*Gerris* sp.) (Hayes and Tennant, 1985). A preference for particular prey species was not observed in this study, and CRLFs appeared to select prey based on availability (Hayes and Tennant, 1985). The largest prey items consumed by large CRLFs (snout-vent length (SVL) >10 cm) were Pacific tree frogs (*Hyla regilla*) and California mice (*Peromyscus californicus*). In this study, vertebrates accounted for over half of the prey mass of larger frogs (Hayes and Tennant, 1985). The study observed juveniles (SVL ≤6.5 cm) feeding day and night. The adult and sub-adult frogs (SVL >6.5 cm) feed only at night.

Observations made during the study suggested that predatory instincts are triggered by movement (Hayes and Tennant, 1985). This led the authors to conclude that CRLFs are not good at identifying prey and tend to forage in an indiscriminant manner (Hayes and Tennant, 1985). The study did not make an effort to observe CRLFs foraging underwater and the prey observed in gut analyses suggest that limited feeding occurs underwater. However, similar studies for ranid frogs have observed the consumption of fish, thus this forage item should not be disregarded (Hayes and Tennant, 1985).

3.4.5.3 Reproduction

California red-legged frogs breed from November to March, with most egg laying occurring in March (FWS, 2002a). Breeding typically occurs during or shortly after major rainfall events (Hayes and Miyamoto, 1984).

Males arrive at breeding sites 2 to 4 weeks prior to females and call as individuals or groups of 2 to 7 frogs (Storer, 1925; FWS, 2002a). Breeding usually occurs in still to slow-moving water greater than 0.7 meters in depth and near dense shrubby riparian vegetation (Hayes and Jennings, 1988). The eggs are laid on emergent vegetation such as bulrushes, cattails, roots, and twigs (Hayes and Miyamoto, 1984). The time to egg hatching depends on water temperature and generally takes 6 to 14 days (Jennings, 1988). Eggs take 20 to 22 days to develop to tadpoles and then 11 to 20 weeks to develop into

terrestrial frogs (Bobzien *et al.*, 2000; Storer, 1925; Wright and Wright, 1949). Males and females reach sexual maturity in 2 and 3 years, respectively, and adults can live up to 10 years (FWS, 2002a).

3.5   Action Area

Aldicarb is highly soluble, mobile in soil, and has a relatively short half-life in soil and water (Kidd and James, 1991; HSDB, 2003; Cohen, 1986). Transport to terrestrial and aquatic environments occurs via surface runoff and subsurface interflow. Volatilization from soil and surface waters is a minor route of transport because aldicarb is applied as a granule that is incorporated into the soil. The granular formulation of TEMIK®, and its immediate incorporation into soil upon application and low Henry's Law constant limit the potential for atmospheric transport.

At present, it is difficult to precisely define the action area for aldicarb in California because of the lack of land use information for some counties. Thus, the formal definition of the action area will, by necessity, rely on a textual description. That is, the action area for aldicarb includes: (1) those areas in California with crops to which aldicarb may be applied according to the pesticide label (Table 3-4), and (2) those areas in California to which aldicarb could be transported following application. The transport of aldicarb and the toxic metabolites aldicarb sulfoxide and aldicarb sulfone will be limited to downstream movement from the point of application. The short half-life of aldicarb and the toxic metabolites in water reduces the distance downstream that they will persist. The physical-chemical properties of aldicarb and the toxic metabolites and the application methods used with TEMIK® eliminate the potential for atmospheric transport to adjacent areas.

3.6   Routes of Exposure

3.6.1   Routes of Transport

Based on the physical-chemical properties and environmental fate of aldicarb and the toxic metabolites, the most likely routes of transport to aquatic and terrestrial life stages of the California red-legged frog, their prey, and their habitat are via surface runoff, subsurface interflow and groundwater discharge. Volitilization of aldicarb and the toxic metabolites from soil surfaces is a minor route of transport and is limited by the immediate incorporation of aldicarb into the soil upon application. Atmospheric transport of aldicarb and the toxic metabolites is an unlikely route of exposure because TEMIK® is formulated as a granule and is immediately incorporated into the soil upon application. Aldicarb and the toxic metabolites are not volatile from surface waters and are not expected to bind to sediments. The reported half-lives for aldicarb and the toxic metabolites in aerobic soil and aquatic environments are relatively short. Aldicarb and the toxic metabolites are considered to have a low bioconcentration potential and do not accumulate in tissues over long-term exposures (Howard, 1991; Smith, 1992; Johnson and Finley, 1980; WHO, 1991; EPA, 2006a). The following sections describe the most likely routes of exposure to aldicarb and the toxic metabolites for terrestrial and aquatic-

phase California red-legged frogs, their prey, and their habitat given the information that has been reported in the previous sections.

3.6.2 Aquatic-Phase California Red-Legged Frog

The two most important routes of exposure for the aquatic life stage of the CRLF (i.e., larvae and tadpoles) are direct exposure to freely-dissolved aldicarb in the water column and ingestion of algae and aquatic plants that contain aldicarb residues. Effects to algae and aquatic plants resulting from exposure to aldicarb are highly unlikely because: 1) the mode of action of aldicarb (i.e., acetylcholinesterase inhibition) does not affect plants (see Section 2.7), and 2) available studies have reported very low toxicity for aldicarb to plants (EPA, 1986). Thus, there is no concern that aldicarb could indirectly affect aquatic-phase CRLFs via reductions in food or aquatic- and terrestrial-phase CRLFs via reductions in habitat availability. Exposure of CRLFs to aldicarb and the toxic metabolites via the consumption of algae and aquatic plants is considered an insignificant exposure pathway given the low K_{ow} (13.5 ml/g) and bioconcentration factors of these compounds.

Based on the high water solubility and mobility of aldicarb in soil, the most likely routes of transport of aldicarb to nearby surface waters are via surface runoff, subsurface interflow, and groundwater discharge. Groundwater discharge is a minor route of transport because of short half-life of aldicarb in aquatic systems, and the slow transport typical of groundwater. Thus, the effects determination for aquatic-phase CRLF focused on exposure of California red-legged frogs, their prey and habitat by direct contact in water (e.g., gills and skin). Exposure of aquatic-phase CRLFs and other biota to aldicarb in sediment and pore water was not estimated because aldicarb is not expected to occur at elevated concentrations in sediment given its physical-chemical properties and fate and behavior characteristics.

Ingestion of TEMIK® granules by aquatic-phase CRLFs and their prey was not considered to be an exposure pathway because: 1) aldicarb is highly water soluble and will not be transported to aquatic systems as granules, and 2) it is applied directly to the field and not to the aquatic environment. Direct application of aldicarb to aquatic environments (e.g., farm ponds, streams) is not permitted, as specified on the product labels.

3.6.3 Terrestrial-Phase California Red-Legged Frog

Aldicarb applied to a field can be transported to terrestrial-phase CRLFs, their prey and habitat by several exposure pathways. Routes of potential exposure for adult CRLFs and their prey include direct contact with aldicarb in the water column (e.g., gills and integument), ingestion of water, granules or contaminated prey, dermal contact and inhalation. Plants in soils treated with aldicarb or in areas receiving run-off from treated fields could be exposed through the uptake of soil pore water, as aldicarb is designed to be adsorbed by roots and transported throughout the plant.

The effects determination for terrestrial-phase CRLF focused on the direct contact (e.g., gills and integument) and ingestion routes of exposure. Terrestrial-phase CRLFs spend

most of their time along shorelines and in aquatic environments. Thus, exposure from direct contact with surface waters is a potential route of exposure. The major route of exposure is also via the respiratory surface (gills) and integument for other freshwater vertebrate and invertebrate prey species. The effects determination focused on CRLFs inhabiting shorelines and aquatic environments. Risks to upland and migratory CRLFs were not specifically considered in the effects determination for the following reasons:

- Over 75% of the individuals monitored by Bulger *et al.* (2003) traveled short distances to upland areas following rain events, but returned to aquatic habitat after a short period, and 90% of these individuals remained within 60 meters of water at all times (Bulger *et al.*, 2003). Upland movements are intermittent and short in duration (4 to 6 days), thus reducing the potential for exposure to aldicarb in upland environments. Given that the primary routes of transport for aldicarb and the toxic metabolites are through surface water runoff and subsurface interflow, CRLFs inhabiting shoreline environments have a higher potential for exposure.

- Migration generally occurs during the summer, well after aldicarb has been applied. Given the physical-chemical properties of aldicarb, granules will no longer be available on the soil surface in the summer and any residues present in prey following application will have been eliminated. Thus, migrating CRLFs are not expected to come in contact with aldicarb granules or contaminated prey.

- The most likely routes of transport of aldicarb and the toxic metabolites are through surface runoff and subsurface interflow to aquatic environments. These routes of transport are not common in upland summer habitats which include small mammal burrows and moist leaf litter, the underside of boulders, rocks, and debris, various agricultural features, and cracks in the bottom of dried ponds (Jennings and Hayes, 1994; FWS, 2002a). With the exception of agricultural features, aldicarb is not applied in these summer habitats. Aldicarb granules are not expected to be available on the soil surface during the summer because they are applied during the spring and quickly dissolve upon contact with water. In the unlikely event that granules are available, CRLFs and their mammalian prey are unlikely to consume granules for the reasons given below.

- In their assessment of California red-legged frog movements and conservation strategies, Bulger *et al.* (2003) suggested that adequate protection of CRLFs can be accomplished by maintaining suitable habitat within 100 meters of aquatic sites. They noted that protecting CRLFs migrating through forests and rangelands is probably unnecessary because it involves <25% of the adult population sporadically moving over large expanses of ubiquitous habitat.

Terrestrial adult CRLFs could be exposed to aldicarb via ingestion of these vertebrate and invertebrate species. It is unlikely that terrestrial vertebrate prey species of CRLFs (*Hyla regilla* and *Peromyscus californicus*) will be directly exposed to aldicarb granules. Pacific tree frogs (*H. regilla*) inhabit areas near water (e.g., springs, ponds, streams, swamps) or in moist environments (e.g., wells, rotting logs, burrows) (Owen, 2000; Morey, 2005). Stebbins (1985) reported that they also inhabit grasslands, chaparral, woodlands, forests, deserts, and farmlands. Resident Pacific tree frogs have an average

home range of 33 m and migratory frogs can range up to 400 m (Morey, 2005; Appendix A). Based on the habitat and home range preferences, it is unlikely that the Pacific tree frog would frequent aldicarb treated fields. The more likely routes of exposure for this species are similar to the CRLF, i.e., through direct contact with integument and ingestion of contaminated prey.

Mice are not expected to be exposed to aldicarb through the consumption of unincorporated and intact granules or by ingesting dissolved aldicarb from pooled water on the soil surface, because:

- If granules are left on the surface following application, it is unlikely that mice would mistake them for seeds. Aldicarb granules are round, black and shiny, and typically range in size from 0.4 to 1.1 mm (Best and Gionfriddo, 1994; Best, L. personal communication, 2005; Uceda and Le Gren, 2001a,b). Most of the weed seeds found in the United States are larger than aldicarb granules and colored differently. Of the nearly 200 weed species seeds described in USDA (1971), approximately seven are similar to aldicarb granules in both color and size. Only three of these weed species – *Portulaca oleracea, Amaranthus retroflexus* and *Spergula arvensis* – are found in agricultural environments (Appendix B).

- For most uses, aldicarb is applied only once per year, thus exposure will be short-term.

- Over 95% of the aldicarb applied in the United States results in >99% incorporation of the granules into the soil surface (Table 3-7). This drastically reduces the granules that are available for consumption.

- The maximum home range radius reported for the California mouse is 35 m (Appendix A). Thus, any mice that consume aldicarb granules are unlikely to be consumed by CRLFs because their home range from treated fields would rarely overlap with CRLFs. Thus, CRLFs are unlikely to forage or be dependent on aldicarb-contaminated mice.

- Mice reach sexual maturity in 5 to 8 weeks. In general, female mice reproduce up to 8 times in their lifespan, with litters averaging 4 to 7 pups. After giving birth, they can become pregnant again in as little as 24 to 48 hours. Because mice reproduce at a high rate and because most of the CRLF diet will come from the nearshore area, it is unlikely that pesticide effects to mice on-site will translate into reduced prey abundance for CRLFs.

- No changes in local rodent and small mammal populations were noted in a series of field studies following use of aldicarb on sugar beets and potatoes in England (Baron and Merriam, 1988). Among the species studied were mice, shrews, voles, and rats.

Therefore, effects to mammaliam prey of CRLFs will not be considered further in this effects determination.

The water balance of frogs is complex, in part because they can absorb water through their skin as well as drink water and extract water from their food (Duellman and Trued,

1986; Minnich, 1982). Some sources report that frogs absorb all of the water that they need through their skin (USDA, 2007). The relative contribution of the different routes of water intake depends on the species, habitat, temperature and body surface area. As a result of this complexity, allometric equations relating body weight and drinking water rates are not available for amphibians (EPA, 1993). Thus, the drinking water route of exposure could not be assessed in the screening-level effects determination.

Table 3-7.   Summary of incorporation efficiencies and frequency of use for TEMIK® 15G brand aldicarb application methods.

Application Method	Incorporation Efficiency		Frequency of Use in
	Bayer Test Results[1]	EPA Default Values[2]	the United States[3]
In-furrow, Drill or Shank	99.96 – 100%	99%	92.5%
Spot Treatment or Injection	No Data	99%	<0.05%
Band – In-furrow Cover with soil to at least 1 inch depth.	100%	99%	3.75%
Band – T-Band	89%	85%	3.0%
Band – Light Incorporation	No Data	85%	0%
Band – Over the Top, Post Emergent	No Data	85%	0.75%
Broadcast – Bed, Row Forming	No Data	85%	0%

Notes:
[1] From Bayer CropScience (1988)
[2] From EPA (2005)
[3] From Hall et al. (2005)

California red-legged frogs could be exposed to aldicarb and the toxic metabolites through the inadvertent ingestion of sediment, soil and sand while foraging in surface water or on land. The incidental ingestion of a considerable amount of sand was observed by a CRLF that consumed a mouse (Hayes and Tennant, 1985). Thus, incidental soil and sediment ingestion is a plausible route of exposure to CRLFs, although it is likely to be less important than food ingestion, given that food ingestion rate far exceeds soil and sediment ingestion rates. Inhalation is not considered a route of exposure given the low potential for total aldicarb residues to volatilize and the immediate incorporation of the pesticide into the soil surface.

Dermal contact is a potential route of exposure for CRLFs and their prey that come in contact with aldicarb granules on the soil surface or burrow in soils containing aldicarb. Less than 1 to 15% of applied aldicarb granules are expected to be available at the soil surface, and exposure to these small amounts via dermal contact is expected to be limited. California red-legged frogs are unlikely to frequent agricultural fields where aldicarb is applied further decreasing the likelihood that dermal contact will be an important route of exposure. CRLF prey, such as mice, that burrow in treated fields could be exposed to aldicarb by dermal contact. However, as noted previously, CRLFs are not expected to come in contact with mice that have been exposed to aldicarb. Thus, potential dermal

exposure of mice to aldicarb is of no significance to CRLFs. Despite the fact that CRLFs are unlikely to come in contact with mice from treated fields, a dermal contact equation was used to determine exposure of mice to granular aldicarb (Hope, 1995; Appendix C). The results suggest that this route of exposure is unlikely to pose a risk to burrowing mammals. Thus, if CRLFs did rely on mice from treated fields as a prey item they would not experience indirect effects from a loss of small mammalian prey.

Therefore, for terrestrial-phase CRLFs the effects determination focused on the direct contact (e.g., gills and integument) and ingestion routes of exposure. Terrestrial-phase CRLFs spend most of their time in and around aquatic environments where they could directly contact aldicarb in surface waters. The same route of exposure exists for their aquatic invertebrate and vertebrate prey, as well as for their terrestrial vertebrate prey, Pacific tree frog. Terrestrial-phase CRLFs feed on a variety of aquatic and terrestrial species that could contain residues of aldicarb. Thus, dietary ingestion is also considered a potential route of exposure for terrestrial-phase CRLFs, as well as for their aquatic and terrestrial vertebrate prey (i.e., fish and Pacific tree frogs).

3.7 Toxicity and Mode of Action

The primary mode of aldicarb toxicity is cholinesterase (ChE) inhibition. The inhibition of the enzyme acetylcholinesterase (AChE) results in the buildup of neurotransmitter acetylcholine (ACh) at cholinergic nerve endings, causing continual nerve stimulation that can result in death (Risher et al., 1987; WHO, 1991). Although the toxic effects of aldicarb are associated with the inhibition of AChE activities in the central nervous system (CNS), depressed AChE activities may also occur in the plasma, muscle, and brain of birds, fish and aquatic invertebrates; and additionally in the red blood cells of mammals (Risher et al., 1987; Perkins and Schlenk, 2000). The inhibition of blood AChE is often used as an indicator of potential AChE inhibition in the CNS (Risher et al., 1987). Exposure to toxic levels of aldicarb can result in the following symptoms: hyperactivity, lethargy, body paralysis, scoliosis, loss of equilibrium, opercular and mouth paralysis, and death (WHO, 1983). Recovery from cholinergic effects is simultaneous to the removal of aldicarb, unless death has already occurred.

Bioconcentration and biomagnification of aldicarb through the food chain are not considered significant exposure pathways (Howard, 1991; Smith, 1992; Johnson and Finley, 1980). Aldicarb has not been found to accumulate in the body during long-term exposures (WHO, 1991). Teratogenic, mutagenic, carcinogenic, or immunotoxic effects have not been associated with aldicarb.

Aldicarb quickly degrades to the metabolites aldicarb sulfoxide and aldicarb sulfone. Aldicarb sulfoxide is a more potent AChE inhibitor than is aldicarb. Results of aldicarb sulfoxide toxicity tests are comparable to that of parent aldicarb in several species. As a result, aldicarb sulfoxide is at least partially responsible for the acute toxic effects resulting from aldicarb exposure. Aldicarb sulfone is far less toxic than parent aldicarb and aldicarb sulfoxide (Baron and Merriam, 1988).

3.8 Ecological Effects

The following section provides an overview of the toxicity of the active ingredient aldicarb and its formulations to aquatic and terrestrial biota. Effects data for amphibians are limited; therefore birds were used as surrogate species for terrestrial-phase CRLFs and fish species were used to assess potential direct effects to aquatic-phase CRLFs, as outlined in EPA (2004b). Aldicarb is not expected to have any adverse effects on terrestrial or aquatic plants at recommended application rates. Its mode of action does not target plants and plants do not possess the enzyme that is inhibited by aldicarb. Thus, plants were not considered in the effects determination for CRLFs exposed to aldicarb. Given that CRLFs depend on aquatic and terrestrial vertebrates and invertebrates for food, toxicity information for these groups was considered in the effects determination. Appendix D provides a more detailed review of the toxicity data available for aquatic and terrestrial biota.

3.8.1 Aquatic Biota

Fish and Invertebrates

Wide ranges of effects values have been reported for freshwater fish and invertebrates. Lethal acute toxicity values for freshwater fish exposed to aldicarb range from 96 hr LC50 values of 49 µg/L for stickleback (*Gasterosteus aculeatus*) to 45,000 µg/L for channel catfish (*Ictalurus punctatus*) (McElligott, 1999; Schlenk, 1995). Studies on aldicarb toxicity have included a variety of freshwater fish species including bluegill sunfish (*Lepomis macrochirus*), rainbow trout (*Oncorhynchus mykiss*), fathead minnow (*Pimephales promelas*), channel catfish, and others. The most frequently studied freshwater fish species have been bluegill sunfish, rainbow trout, and fathead minnow. The most sensitive chronic endpoint reported in the literature is for the survival of early life stages of fathead minnows exposed to aldicarb (Pickering and Gilliam, 1982). This study reported a NOAEC (no observed adverse effects concentrations) of 78 µg/L and a LOAEC (lowest observed adverse effects concentrations) of 156 µg/L following 30 days exposure of larvae-juvenile fathead minnows to aldicarb.

Values for the acute toxicity of aldicarb to freshwater invertebrates range from a 48 hr LC50 of 20 µg/L for *Chironomus tentans* to a 96 hr LC50 of >320,000 µg/L for Asiatic clam (*Corbicula fluminea*) and mussels (*Elliptio complanata*) (Moore et al., 1998; Moulton et al., 1996). Studies examining toxicity of aldicarb to aquatic plants and algae are limited. Those available report a low concern for acute toxicity (EPA, 1986). The bioconcentration of aldicarb in aquatic biota is low (Howard, 1991; Smith, 1992; Johnson and Finley, 1980). Aldicarb has not been found to accumulate in the body during long-term exposures (WHO, 1991).

Metabolites

Aldicarb sulfoxide is less toxic to freshwater fish species than the parent compound, aldicarb. Acute toxicity values reported for bluegill and rainbow trout range from a 72 hr LC50 of 4,000 to a 96 hr LC50 of 7,440 µg/L, respectively (Clarkson and Hensley, 1968; Odin-Fuertet, 1998). Acute toxicity values reported for freshwater invertebrate species exposed to aldicarb sulfoxide are similar to those reported for aldicarb. Values reported for adult *Daphnia laevis* range from a 48 hr EC50 of 43 µg/L to an adult 48 hr LC50 of

103 µg/L (Foran *et al.*, 1985). *Daphnia magna* are less sensitive with 72 hr LC50 values ranging from 696 to 800 µg/L (Handley *et al.*, 1995; MRID 45592117).

Acute toxicity values reported for freshwater fish species exposed to aldicarb sulfone are higher than those for freshwater fish exposed to aldicarb. Rainbow trout and bluegill sunfish exposed to aldicarb sulfone were observed to have 96 hr LC50 values of 42,000 and 53,000 µg/L, respectively (Union Carbide Corporation, 1975). Foran *et al.* (1985) reported acute toxicity values for *Daphnia laevis* ranging from a 48 hr EC50 of 369 to a 48 hr LC50 of 1,124 µg/L. Similar toxicity values were reported for *Daphnia magna*, with 48 hr LC50 values of 280 and 550 µg/L (Union Carbide Corporation, 1975; Handley *et al.*, 1994).

Aldicarb sulfoxide and aldicarb sulfone are either less toxic or have similar toxicity than the parent compound depending on the tested species.

3.8.2 Terrestrial Biota

Birds

Similar ranges in toxicity are reported in the literature for different bird species. Orally administered acute LD50s range from 0.75 mg/kg for common grackle (*Quiscalus quiscula*) and house sparrow (*Passer domesticus*) to 9.5 mg/kg for white leghorn cockerel (*Gallus gallus*) (Schafer and Brunton, 1979; West and Carpenter, 1965). Studies have been performed using a number of bird species, including: northern bobwhite (*Colinus virginianus*), valley quail (*Lophortyx californicus*), Japanese quail (*Coturnix coturnix japonica*), California quail (*Callipela californica*), mallard duck (*Anas platyrhynchos*), and mourning dove (*Zenaida macroura*).

Invertebrates

Toxicity data for terrestrial invertebrates other than earthworms are lacking. Earthworms are not part of the California red-legged frog diet (see Section 4.1.3), and thus were not considered in the effects determination. The potential for effects to beneficial and non-target insects are considered negligible based on the results of several field studies (Baron and Merriam, 1988). These field studies were conducted in the Delta States, Texas and California and involved observation of beneficial insects for adverse effects following the application of TEMIK® (Baron and Merriam, 1988). Some short-term declines were reported in treated fields. The EFED assessment for aldicarb concluded that the pesticide is toxic to honey bees. However, studies exposing honey bees to aldicarb applied to orchards at 5 and 10 lbs ai/A and field crops at 3 and 4 lbs ai/A did not observe significant impacts on bee survival, pollination efficiency, or colony vigor (Baron and Merriam, 1988). Although aldicarb is toxic to these insects as a contact poison, the likelihood of population-level impacts resulting from the labeled use of TEMIK® granules is negligible (Baron and Merriam, 1988). Over 99% of the TEMIK® granules that are applied are incorporated into the soil where the dissolve upon contact with water and are systemically absorbed by plants.

Metabolites

The acute toxicity of aldicarb sulfoxide is similar to that of the parent compound, but is lower for aldicarb sulfone. Orally administered toxicity studies are not available for birds and aldicarb sulfoxide. An acute dietary LD50 of 362 mg/kg bw was reported for northern bobwhites exposed to aldicarb sulfoxide (Gallagher *et al.*, 1998a). No studies were found that exposed northern bobwhites to aldicarb in feed. Dietary studies have been performed with common pheasant and Japanese quail. Acute dietary LD50s of >300 mg/kg and 247 to 786 mg/kg were reported for common pheasant and Japanese quail, respectively (Hill *et al.*, 1975; Hill and Camardese, 1981). Fink (1976a,b) reported dietary and oral acute LD50 values of >10,000 and 33.5 mg/kg, respectively, for mallard ducks exposed to aldicarb sulfone. The oral LD50 for rats exposed to aldicarb sulfoxide and aldicarb sulfone are 0.88 and 25 mg/kg bw, respectively (WHO, 1991; Baron, 1994).

3.8.3 Acute Versus Chronic Exposure

Several studies have shown that most mortality occurs in the first 24 to 48 hours of a bioassay (Thun, 1990a; EPA, 1981). As a result, LC50s and other effects endpoints do not change much after the initial 24 to 48 hours of the bioassay. Thus, chronic exposure (>96 hr) to aldicarb is unlikely to result in significant additional mortalities. Additional reasons why chronic toxicity was not considered a major concern include:

- Aldicarb is a fast-acting cholinesterase (ChE) inhibitor. Effects are reversible shortly after the removal of aldicarb (Risher *et al.*, 1987).

- The aquatic half-life of aldicarb is short (i.e., 3-6 days), thus chronic exposure to aquatic organisms is unlikely to occur (Baron, 1991; MRID No. 45592107).

- Aldicarb is rarely applied more than once a season (the exceptions are cotton and sugar beets), and when it is applied more than once, applications are at least 4 days apart (Hall *et al.*, 2005).

- It is not persistent and does not bioconcentrate (Johnson and Finley, 1980; Howard, 1991; WHO, 1991; Smith, 1992).

- Acute and chronic studies report similar levels of toxicity, thus acute effects are likely to supersede chronic effects (see Appendix D).

Despite the fact that adverse effects from chronic exposure are unlikely to occur, the effects determination for California red-legged frogs did consider this exposure duration.

3.9 Conceptual Model

The conceptual model provides a written and visual description of the possible exposure routes between ecological receptors and a stressor. The model includes risk hypotheses for how a stressor might come in contact with, and affect, receptors at a site. Risk hypotheses are derived using professional judgment and information available on the sources of exposure, characteristics of the stressor (e.g., chemistry, fate and transport), the ecosystems at risk, and anticipated effects to ecological receptors.

3.9.1 Risk Hypotheses

Risk hypotheses are specific assumptions about potential adverse effects (i.e., changes in assessment endpoints) and may be based on theory and logic, empirical data,

mathematical models, or probability models (EPA, 1998). For this assessment, the risk is stressor-linked, where the stressor is the release of aldicarb to the environment. Based on the results of the EFED risk assessment for aldicarb (EPA, 2006a), the following risk hypotheses are put forth for this effects determination:

- Aldicarb in surface water and/or runoff from treated areas may directly affect CRLFs by causing mortality, or adversely affecting growth or reproduction;

- Aldicarb in surface water and/or runoff from treated areas may indirectly affect CRLFs by reducing or changing the abundance and composition of aquatic and terrestrial prey populations; and

- Aldicarb in surface water and/or runoff from treated areas may indirectly affect CRLFs by reducing or changing the composition of the aquatic and terrestrial plant communities in CRLF habitat, thus affecting primary productivity and/or cover.

3.9.2 Diagram

Figure 3-3 presents the conceptual model for evaluating risks to the aquatic and terrestrial life stages of the California red-legged frog from the use of aldicarb. The conceptual model shows the anticipated sequence of events following application of aldicarb.
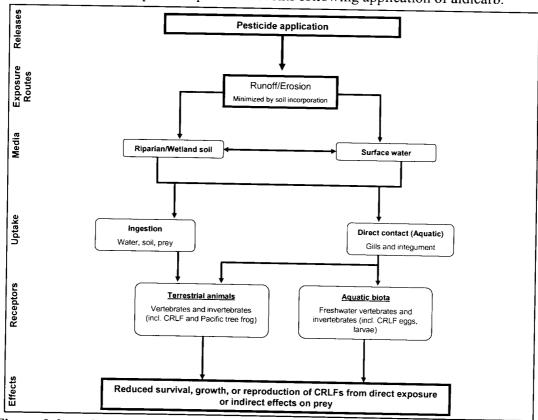

Figure 3-3. Conceptual model for the application of aldicarb in California, leading to exposure of California red-legged frogs and their prey.

Aldicarb and the toxic metabolites may be transported to nearby aquatic systems via surface runoff, subsurface/interflow, and groundwater discharge. The potential for surface runoff is reduced by the incorporation of aldicarb into the soil during application, thus minimizing the number of aldicarb granules left on the soil surface. Movement of aldicarb to groundwater is generally rare. The potential for groundwater contamination is elevated in areas with coarser soils, soils and subsoils with low organic matter, shallow water tables, and rainfall and irrigation that exceed evapotranspiration (Coppedge et al., 1977; Bowman, 1988; Cohen et al., 1986). However, risk of groundwater contamination has been reduced by geographical prohibitions, soil restrictions, rate reductions and timing of applications that have been introduced to protect vulnerable drinking water sources. Aldicarb use patterns were frequently revised between 1980 and 2003 to achieve this. The most significant label changes were in 1989 and included well head protection (50 foot buffer zones for all wells) and well setback restrictions for shallow wells by soil type, water table depth, and depth of well casing. These changes were supplemented by the adoption of a best management plan on the Florida ridge (Florida, 1994) and an active stewardship program (Hovis, 2005). Recent targeted monitoring programs in vulnerable areas have confirmed the effectiveness of these measures (Lenz et al., 2006; Wyatt et al., 2006; Freeseman et al., 2006; Mosier et al., 2006; Lee et al., 2006). Any detectable residues were low in concentration with none exceeding EPA drinking water standards (EPA, 2004a). Monitoring studies have shown that groundwater discharge is not a significant transport mechanism for aldicarb residues to surface water (Jones and Estes, 1995). The rapid degradation of aldicarb in stream beds under anaerobic conditions is likely an important process limiting this route of transport (Jones and Estes, 1995). The dilution of groundwater in surface water bodies also reduces the concentration of aldicarb residues not degraded in the sediment layer.

Uptake through the gills and integument of aquatic organisms and ingestion of prey containing aldicarb residues were considered the primary modes of uptake for aquatic-phase CRLFs and their prey. The most likely routes of exposure for terrestrial CRLFs and their prey are through direct contact and ingestion of granules, water or prey containing aldicarb residues. Incorporation of aldicarb into the soil surface during application minimizes the number of granules on the soil surface, thus reducing the potential for exposure by this route. Inhalation and dermal contact are considered minor routes of exposure for terrestrial CRLFs and their prey. Based on the physical and chemical properties of aldicarb (Table 3-5), bioconcentration and biomagnification through the food chain were not considered significant exposure pathways.

3.10 Protection Goals and Assessment Endpoints

Protection goals are defined by scientific knowledge and societal values. They describe the overall aim of a risk assessment or effects determination and are used as the basis for defining assessment endpoints. In turn, assessment endpoints are ecological characteristics that are deemed important to evaluate and protect (e.g., survival of California red-legged frogs). They guide the assessment by providing a basis for assessing potential risks to receptors. Factors considered in selecting assessment endpoints include mode of action, potential exposure pathways, and sensitivity of ecological receptors. Assessment endpoints can be general (e.g., bird reproduction) or

specific (e.g., nesting success of red-winged blackbirds) but must be relevant to the ecosystem they represent and susceptible to the stressors of concern (EPA, 1998).

Section 7(a)(2) of The Endangered Species Act, and implementing regulations consistently indicate that the protection goal with respect to listed species potentially exposed to pesticides is the jeopardy of the continued existence of listed species or destruction or adverse modification of their habitat. Therefore, the protection goal for the California red-legged frog is to ensure that exposure to aldicarb is not likely to jeopardize the continued existence of the California red-legged frog, result in the destruction or adverse modification of the habitat of this species, or cause indirect effects to prey the CRLF depends on. For direct toxic effects to the California red-legged frog, the starting assessment endpoint is the survival, reproduction and growth of this species. An organism-level assessment endpoint is used for the assessment of direct toxic effects to the California red-legged frog.

The following assessment endpoints were chosen to address indirect effects of aldicarb to the California red-legged frog:

- Primary productivity of the algal community in aquatic environments that contain or potentially contain early life stages of the California red-legged frog. Early life stages of California red-legged frog are algal grazers and thus require that an abundance of this prey item be maintained. This assessment endpoint is at the community level of organization because it is unlikely that CRLFs would graze solely on a few sensitive species of algae.

- Productivity of invertebrates and small vertebrates associated with aquatic and terrestrial habitats of adult California red-legged frogs. Adult California red-legged frogs forage opportunistically on a variety of invertebrate and vertebrate prey in or near their preferred aquatic habitats. This assessment endpoint is at the community level of organization because it is unlikely that adults forage solely on a few sensitive invertebrate or vertebrate species.

- Structure of the plant community in the near-shore environments that contain or potentially contain early and adult life stages of the California red-legged frog. For the purpose of this assessment, the terrestrial habitat of the CRLF consists of all upland habitats within 100 meters of aquatic sites. The assessment endpoint for habitat is at the community level of organization because it is unlikely that the absence of one or a few sensitive plant species would adversely affect the habitat of the California red-legged frog.

In addition to the need to have a general assessment endpoint for indirect effects to CRLF habitat, there is a need to have assessment endpoints for CRLF critical habitats, as defined by the Fish and Wildlife Service (FWS, 2006). Critical habitat is defined in Section 3 of the Endangered Species Act as: (i) the specific areas within the geographical area occupied by the species...on which are found those physical and biological features essential to the conservation of the species and that may require special management

considerations or protection, and (ii) specific areas outside the geographical area occupied by a species at the time it is listed, upon a determination that such areas are essential in the conservation of the species. In determining which areas to designate as critical habitat, FWS considers those physical and biological features (PCEs) that are essential to the conservation of the California red-legged frog. The FWS final rule on critical habitat for the CRLF took effect on May 15, 2006. Approximately 450,228 acres of critical habitat has been designated for the California red-legged frog (Figures 3-4a,b,c) (FWS, 2006). Critical habitat is either occupied by the CRLF, is within the historic range of the CRLF, and/or contains sufficient primary constituent elements (PCE) to support at least one life history function of the CRLF. Primary constituent elements are physical and biological features that are considered essential to the conservation of the CRLF. Four PCEs have been identified that represent the life history functions of the CRLF: aquatic breeding habitat, aquatic non-breeding habitat, upland habitat, and dispersal habitat (see text box for descriptions).

Because of the special concern associated with protection of critical habitats of the California red-legged frogs, the following assessment endpoints were developed for each of the primary constituent elements of CRLF critical habitats:

- Community structure of the plant community that constitutes aquatic breeding habitat of the California red-legged frog.

- Community structure of the plant community that constitutes aquatic non-breeding habitat of the California red-legged frog.

- Community structure of the plant community that constitutes upland habitat of the California red-legged frog.

- Community structure of the plant community that constitutes dispersal habitat of the California red-legged frog.

The PCE assessment endpoints for critical habitat are at the community level of organization because it is unlikely that the absence of one or a few sensitive plant species would lead to adverse effects to the California red-legged frog.
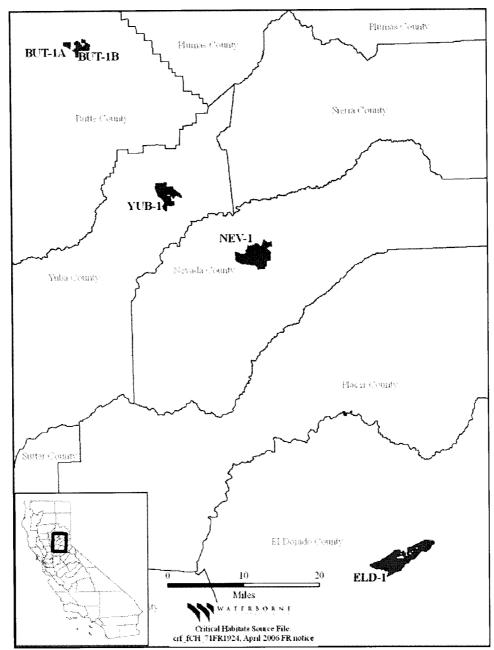
Figure 3-4a. Critical habitat for the California red-legged frog in northern California.
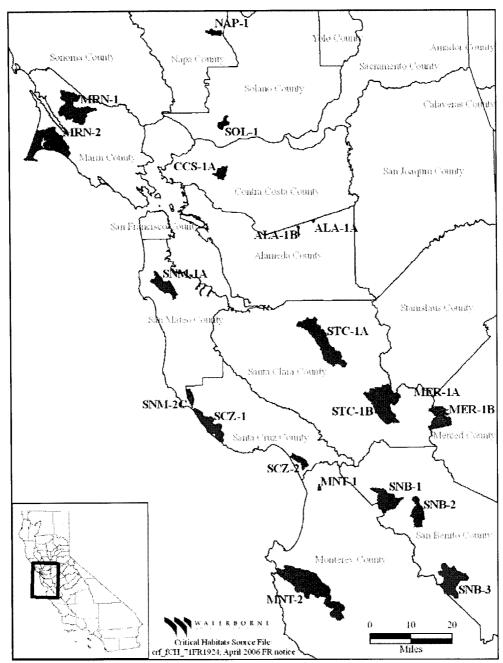
Figure 3-4b. **Critical habitat for the California red-legged frog in central California.**
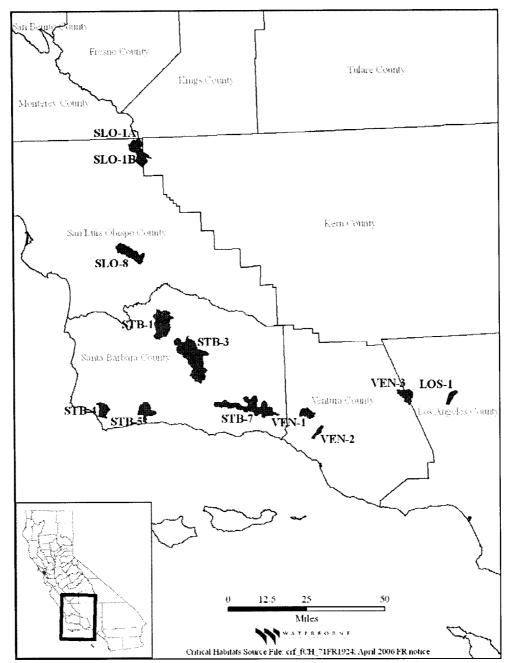
Figure 3-4c. **Critical habitat identified for the California red-legged frog in southern California.**

3.11 Measures of Exposure and Effects

Measures of exposure and effect are the attributes used to quantify potential risks to an assessment endpoint (EPA, 1998). The challenge for risk assessors is to select measures that will provide sufficient information to evaluate potential risks to the assessment endpoint. EPA (1998) groups measures into one of three categories. Measures of effect

are measurable changes in an attribute of the assessment endpoint, or a surrogate, in response to the stressor. Measures of exposure account for the presence and movement of the stressor in the environment and co-occurrence with the assessment endpoint. Measures of ecosystem and receptor characteristics consider the influence that the environment, organism behavior, and life history will have on exposure and response to the stressor (EPA, 1998). This effects determination for the CRLF builds upon past assessments of aldicarb. Measures of effect (e.g., LC_{50}, NOAEC) were chosen based on those used in the EFED assessment for aldicarb (EPA, 2006a) and on information presented in the problem formulation. Measures of effect from the EFED assessment were not used in cases where a more conservative value was identified in the current literature. In addition, the results from the EFED assessment were summarized and assessment endpoints determined to be of no concern were removed from consideration in the effects determination.

To evaluate the potential risk to non-target organisms from the use of aldicarb products, the EFED assessment (EPA, 2006a) calculated risk quotients (RQs) by dividing estimated environmental concentrations (EECs) by acute and chronic toxicity values (e.g. LC_{50}, NOEC). These RQs were then compared to the Agency's levels of concern (LOCs) to determine whether aldicarb had the potential to cause adverse effects to non-target organisms.

For aquatic organisms, EECs were modeled for the use of aldicarb on cotton, potato, citrus, pecans, and soybeans. The aquatic scenarios were chosen because they represent the major aldicarb use crops and represent a variety of geographic use areas and application rates (Table 3-8). Thus, they were expected to account for other use scenarios as well. The use sites modeled in the aquatic assessment were considered the most vulnerable to pesticide run-off leading to higher aquatic exposures. For the aquatic exposure scenarios, EECs were generated for total aldicarb residues (parent + sulfoxide + sulfone). This was a conservative approach because it assumed that aldicarb and the metabolites elicit the same level of effects on aquatic biota. However, aldicarb is generally more toxic than aldicarb sulfoxide which, in turn, is more toxic than aldicarb sulfone.

Table 3-8. **EFED assessment exposure scenarios used to estimate aldicarb concentrations in the aquatic environment (EPA, 2006a).**

State/Crop	Typical use rate (lbs a.i./A)	Maximum labeled rate (lbs a.i./A)
MS/Cotton	-	4.05
MS/Cotton	0.6	4.05[*]
ID/Potato	2.7	3.0
FL/Citrus	3.7	4.95
MS/Soybean	0.7	3.0
GA/Pecan	3.1	10.05

[*]Total from two applications.
The typical use rate for cotton could not be found.

For terrestrial organisms, EECs were modeled for all of the federally-labeled uses of aldicarb: citrus, cotton, dry beans, sorghum, peanuts, pecans, potatoes, soybeans, sugar beets, sugarcane, sweet potatoes, and ornamentals. The terrestrial exposure scenarios were modeled using maximum and typical application rates, label application methods and parameters (Table 3-9). The metabolites aldicarb sulfoxide and aldicarb sulfone were not included in the results for the terrestrial exposure scenarios because they are unlikely to form in significant quantities in the terrestrial environment (EPA, 2006a). The Agency chose a conservative approach by using the most sensitive toxicity endpoint values for each surrogate taxa (e.g., birds, mammals, fish, invertebrates) in the RQ calculations.

Table 3-9. **Exposure scenarios used to estimate aldicarb concentrations in the terrestrial environment (EPA, 2006a).**

Crop	Application method	Typical use rate (lbs a.i./A)	Maximum labeled rate (lbs a.i./A)	Incorporation efficiency (%)
Cotton	Banded	0.6	4.05	85
Dry beans	Banded	1.0	2.1	85
Sorghum	In-furrow	0.4	1.05	99
Peanuts	Banded	0.9	3.0	85
Potatoes	Banded	2.7	3.0	85
Soybeans	Banded	0.7	3.0	85
Sugar beets	Banded	1.8	4.95	85
Sweet potatoes	Banded	1.4	3.0	99
Citrus	Broadcast	3.7	4.95	85
Pecans	Broadcast	3.1	10.05	85
Ornamentals	Broadcast	Not available	5.0	85

EPA calculated the RQs for fish and aquatic invertebrates using the EECs for the maximum labeled use rates of aldicarb (Table 3-8). The RQs were then compared to the acute risk, restricted use, endangered species and chronic risk LOCs. The acute RQs for fish ranged from 0.03 to 0.54 and for aquatic invertebrates they ranged from 0.07 to 1.40. The results are presented in Table 3-10. The only scenario that did not exceed an acute LOC was the application of aldicarb to potatoes using a typical application rate.

Table 3-10. **Acute risk quotients for fish and aquatic invertebrates reported in the EFED assessment (EPA, 2006a).**

Scenario (lb ai/A, No. of appl.)	Fish LC50 (µg/L)	Invertebrate EC50 (µg/L)	EEC Peak (µg/L)	Acute Risk Quotient Fish	Acute Risk Quotient Invertebrates
Cotton 4.05, 1 appl.	52	20	28.04	0.54***	1.4***
Cotton 4.95, 2 appl.	52	20	18.40	0.35**	0.92***
Potato 3.00, 1 appl.	52	20	1.43	0.03	0.07*
Citrus 4.95, 1 appl.	52	20	2.96	0.06*	0.15**
Soybeans 3.00, 1 appl.	52	20	7.12	0.14**	0.36**

Table 3-10. Acute risk quotients for fish and aquatic invertebrates reported in the EFED assessment (EPA, 2006a).

Scenario (lb ai/A, No. of appl.)	Fish LC50 (µg/L)	Invertebrate EC50 (µg/L)	EEC Peak (µg/L)	Acute Risk Quotient Fish	Invertebrates
Pecans 10.05, 1 appl.	52	20	12.04	0.23[**]	0.60[***]

[*]Exceeds endangered species LOC (0.05).
[**]Exceeds endangered species and restricted use LOC (0.1).
[***]Exceeds endangered species, restricted use and acute risk LOC (0.5).

The chronic RQs for fish and aquatic invertebrates exceeded the chronic LOC of 1 for all of the exposure scenarios. The chronic RQs for fish and aquatic invertebrates ranged from 2.93 to 53.36 and from 1.40 to 26.56, respectively (EPA, 2006a). EPA calculated the chronic fish RQ using an NOAEC value of 0.46 µg/L that was derived from an acute to chronic ratio for fathead minnows (*Pimephales promelas*) and applied to an acute toxicity value for bluegill sunfish (*Lepomis macrochirus*). This value was used instead of the NOAEC of 78 µg/L for fathead minnow (EPA, 2006a). The Agency reasoned that using the fathead minnow value would not account for more sensitive species like bluegill sunfish (EPA, 2006a).

Risk quotients were not calculated for aquatic plants in the EFED assessment because studies in the open literature lacked the quantitative data necessary to calculate risk (EPA, 2006a). These studies indicated a low toxicity to terrestrial plants from exposure to aldicarb. An EC50 of >50,000 µg/L was reported for the marine diatom, *Skeletonema costatum* (EPA, 1986). This value is far greater than all of the modeled acute and chronic concentrations and the peak monitoring value reported in the EFED assessment (EPA, 2006a).

The RQs for birds and mammals exceeded the acute risk, restricted use, endangered species and chronic risk LOCs for all of the terrestrial exposure scenarios modeled in the EFED assessment of aldicarb (EPA, 2006a). Using the RQ approach, the EFED assessment concluded that there is a potential risk to honeybees and other beneficial insects. EPA acknowledged that because aldicarb is formulated as a granular it is unlikely the honeybees will experience direct exposure, but exposure through plants and soil could occur. The assessment also concluded that modeled application rates could pose a risk to earthworms. Due to a lack of quantitative data, the EECs were not compared to toxicity data for terrestrial plants. However, the lack of effects reported in studies examining the exposure of aldicarb to terrestrial plants indicates that aldicarb has a low toxicity to this receptor group.

3.11.1 Measures of Exposure and Effects Removed From Further Consideration

Based on the results of the EFED assessment of aldicarb, the following assessment endpoints and measures of effects were removed from further consideration in the effects determination:

Aquatic-phase California red-legged frog

- Acute and chronic effects to the primary productivity of the algal community in aquatic environments that potentially contain early life stages of the California red-legged frog.

- Acute and chronic effects to the structure of the plant community in aquatic environments that potentially contain early life stages of the California red-legged frog.

- Acute and chronic effects to the structure and function of the aquatic breeding and aquatic non-breeding primary constituent elements (PCE) of critical habitat for the California red-legged frog.

Terrestrial-phase California red-legged frog

- Acute and chronic effects to the structure of the plant community in terrestrial environments that potentially contain adult California red-legged frogs.

- Acute and chronic effects to the structure and function of the upland and dispersal primary constituent elements (PCE) of critical habitat for the California red-legged frog.

- Indirect effects to vegetative habitat from acute and chronic exposure to aldicarb.

The routes of exposure section for adult CRLF (Section 3.6.3) determined that exposure of mice (terrestrial prey) to aldicarb is unlikely based on the methods used to apply aldicarb and the characteristics of the granules. The home range of mice inhabiting treated fields is not expected to overlap with that of the CRLF. Thus, CRLFs would not be dependent on mice potentially exposed to aldicarb. In the unlikely event that exposure was to occur, it is not expected to impact prey abundance to a degree that would adversely affect the CRLF. Indirect effects to mammals were not considered in this ESA.

Although the EFED assessment categorized aldicarb as highly toxic to honey bees, EPA acknowledged that direct contact of honey bees and other beneficial insects is unlikely with a granular pesticide such as aldicarb. However, EPA felt that honey bees and beneficial insects could be exposed to aldicarb in soil and plants given the systemic nature of the pesticide. Numerous studies have reported negligible effects to beneficial and non-target insects exposed to aldicarb (Baron and Merriam, 1988). The potential exposure of insects to aldicarb is limited to those individuals that occur on treated fields. Exposure to these individuals is reduced by the incorporation of aldicarb granules into the soil. Ninety percent of the CRLFs that Bulger *et al.* (2003) monitored stayed with 60 m of water at all times and were almost always within 5 m of their summer aquatic habitat. The furthest upland movement that frogs made was during rain events when they traveled up to 130 m upland for approximately 4 to 6 days (Bulger *et al.*, 2003). In the unlikely event that insect populations on fields treated with aldicarb were reduced, it is not expected to impact prey abundance to a degree that would adversely affect the CRLF. Indirect effects to CRLFs via reduction in abundance of terrestrial insects were not considered in this effects determination.

A chronic toxicity value for birds (used as a surrogate for terrestrial-phase CRLF) was not available for the EFED assessment. EPA states that it does not currently perform

quantitative chronic risk assessments for granular pesticides on terrestrial organisms (EPA, 2006a). It is unlikely that terrestrial CRLFs will experience chronic exposure:

- Aldicarb is typically applied only once per year.

- The granules are incorporated into the soil.

- The granules dissolve upon contact with water.

Therefore, chronic effects to terrestrial-phase CRLFs were not considered in the screening-level effects determination. Chronic effects were considered for aquatic-phase CRLFs and aquatic prey items.

A summary of the assessment endpoints and measures of exposure and effect that were selected to characterize potential risks to CRLFs associated with exposure to aldicarb are provided in Tables 3-11. Effects metrics for prey items of the CRLF will be based on laboratory ecotoxicological data.

Table 3-11. Summary of assessment endpoints and measures of exposure and effect for screening-level CRLF effects determination.

Assessment Endpoint	Exposure Duration	Measures of Exposure	Measures of Ecological Effect
Aquatic Life Stage			
Survival, reproduction and growth of the CRLF.	Acute	Estimated 1 in 10 year peak concentrations of pesticides in standard-sized ponds located adjacent to treated agricultural fields based on the 30-year simulation.	96 hr LC50 for bluegill sunfish.
	Chronic	Estimated 1 in 10 year 21-day average concentrations of aldicarb based on the 30-year simulation in standard-sized ponds located adjacent to treated agricultural fields.	30-day NOEC (growth and survival) for larvae-juvenile fathead minnows.
Terrestrial Life Stage			
Survival, reproduction and growth of the CRLF.	Acute	Total daily intake (TDI) modeling for CRLFs exposed to aldicarb in diet. Estimated 1 in 10 year peak concentrations of pesticides in standard-sized ponds located adjacent to treated agricultural fields based on the 30-year simulation.	24 hr LD50 for mallard duck.
Productivity of invertebrates and small vertebrates associated with nearshore habitats of adult California red-legged frogs.	Acute	Total daily intake (TDI) modeling for Pacific tree frogs exposed to aldicarb in diet. Estimated 1 in 10 year peak concentrations of pesticides in standard-sized ponds located adjacent to treated agricultural	48 hr EC50 for Chironomus tentans. 96 hr LC50 for bluegill sunfish (for fish and surrogate for aquatic-phase amphibian prey). Terrestrial invertebrates not considered (see Section 3.11.1). Mice not considered (see Section 3.6.3).

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Table 3-11. Summary of assessment endpoints and measures of exposure and effect for screening-level CRLF effects determination.

Assessment Endpoint	Exposure Duration	Measures of Exposure	Measures of Ecological Effect
		fields based on the 30-year simulation.	
	Chronic	Total daily intake (TDI) modeling for pacific tree frogs exposed to aldicarb in diet. Estimated 1 in 10 year 21-day average concentrations of aldicarb based on the 30-year simulation in standard-sized ponds located adjacent to treated agricultural fields.	30-day NOEC (growth and survival) for larvae-juvenile fathead minnows. 21-day NOEC (immobilization) for *Daphnia magna*. Terrestrial invertebrates not considered (see Section 3.11.1). Mice not considered (see Section 3.6.3).

3.12 Exposure Scenarios

The exposure scenarios that were evaluated in this effects determination were based on the scenarios that were modeled in the Agency's assessment for aldicarb (EPA, 2006a), the uses reported on the TEMIK® pesticide labels, the most common uses of aldicarb reported for California (Table 3-2) (Cal DPR, 2005), and use restrictions that are specific to the state. The effects determination initially focused on the worst-case exposure scenarios for aldicarb, and only considered other scenarios as the need arose.

The EFED assessment for aldicarb used different exposure scenarios for aquatic and terrestrial biota. For aquatic biota, exposure scenarios included cotton, potatoes, citrus, pecans, and soybeans (Table 3-8). EPA chose these scenarios because they represent the major aldicarb use crops, geographic use areas, and application rates. Exposure to terrestrial animals was modeled for all of the federally-labeled uses of aldicarb: citrus, cotton, dry beans, sorghum, peanuts, pecans, potatoes, soybeans, sugar beets, sugarcane, sweet potatoes, and ornamentals (Table 3-9). With the exception of fish exposed to aldicarb applied to potatoes at a typical application rate, all of the exposure scenarios modeled in the EFED assessment exceeded at least one of the LOCs for fish, aquatic invertebrates, birds, and mammals.

In 2005, the top five uses of aldicarb in California, in terms of active ingredient applied, were for cotton, pecans, soil application (e.g., seedbeds), dried beans, and sorghum/milo (Table 3-2). Cotton accounted for 99% of the use of aldicarb in California in 2005 (Cal PUR, 2005). General application instructions and restrictions specific to California were considered in determining the exposure scenarios that were modeled in the effects determination. Aldicarb is applied only once per year to most crops. The exceptions are cotton and sugar beets which can receive at planting and post-emergence applications (EPA, 2006a). The application of TEMIK® to crops in California is only permitted between March 1 and September 1. Application rates specific to the use of TEMIK® in California are reported for most crops. Where appropriate, these use rates were substituted for national use rates modeled in the EFED assessment for aldicarb. In California, the use of TEMIK® is not permitted on citrus, potatoes and sugarcane, thus these crops were removed from further consideration in the effects determination. In California, the use of aldicarb on pecans is permitted under a special local need registration (No. CA-990026).

Table 3-12 presents a list of exposure scenarios that accounts for the scenarios used in the EFED assessment (EPA, 2006a), the label-permitted uses, application rates, and application methods of TEMIK® in California, and the most common uses of aldicarb reported in the California Pesticide Use Reports (Cal DPR, 2005).

Table 3-12.   Exposure scenarios for the CRLF effects determination for aldicarb.

Scenario	Application Method	Soil	Application Rate (lbs a.i./A)

			Typical Single	Maximum Single	Maximum Seasonal
Cotton	In-furrow at planting	99%	0.600	2.10	3.15
	Band at planting	85%	1.05	2.10	
Peanuts	In-furrow at planting	99%	1.05	2.10	1 appl./crop
	Band at planting	85%	2.10	3.00	
Pecans	In-furrow	99%	2.55	4.95	1 appl./year
	Band and light incorporation	85%	2.55	4.95	
Sugar beets	In-furrow at planting and post-emergence	99%	1.80	2.10 + 2.10	4.20
	Band at planting and post-emergence	85%	1.80	2.10 + 2.10	
	In-furrow at planting	99%	1.05	2.10	
	Band at planting	85%	1.05	2.10	

Table 3-13 lists the worst-case exposure scenarios. They account for the predominant use of aldicarb on cotton, the highest permitted label use rates for ornamentals, and the use of the pesticide on orchard and field crops. The screening-level effects determination began with the worst-case scenarios in Table 3-13. The remaining scenarios in Table 3-12 were modelled only if the worst-case scenario results indicated a need to do so. In keeping with the conservative nature of the screening-level effects determination, the exposure scenarios used the maximum label permitted application rate and the application method that produced the greatest off-site transport of aldicarb. Multiple applications of TEMIK® are permitted with cotton and sugar beets. In the case of cotton, the maximum seasonal application rate of 3.15 lbs ai/A was modeled using two applications of 1.05 and 2.10 lbs ai/A separated by a 28 day interval. Sugar beets were modeled using two applications of 2.10 lbs ai/A separated by a 21 day interval. The first application was modeled for in-furrow application with 99% soil incorporation of the granules and the second application was modeled for the side dressed method with 85% soil incorporation efficiency. Pecans were modeled at the maximum label permitted rate of 4.95 lbs ai/A and 85% soil incorporation of the granules.

Table 3-13.   Exposure scenarios modeled using PRZM/EXAMS.

Crop	Application Method	Soil Incorporation Efficiency	Application Rate (lbs ai/A)
Cotton	In-furrow at planting and side dressed	99%	1.05 + 2.10
Sugar beets	In-furrow at planting and side dressed	99% + 85%	2.10 + 2.10
Pecans	Band application with light incorporation	85%	4.95

3.13 Analysis Plan

A screening-level effect determination was performed to determine if the exposure scenarios in Table 3-13 posed risks to California red-legged frogs or their prey. Based on

the results of the screening-level effects determination further evaluation was determined to be unnecessary. A refined effects determination would have been necessary if potential adverse effect to the CRLF and their prey had been identified for any of the exposure scenarios. The screening-level effects determination was conducted in three phases: exposure assessment, effects assessment, and risk characterization. The following sections describe the approach used to complete each of these phases.

3.13.1 Screening-level Effects Determination

Environmental fate data, monitoring data, and computer models were used to derive estimated exposure concentrations (EECs) for the exposure assessment. Effect metrics were based on sensitive endpoints from high quality studies on relevant species. Risk was estimated by calculating the ratio of the EEC and the corresponding measure of effect. This ratio is the risk quotient (RQ), which was then compared to pre-established acute and chronic levels of concern (LOCs) for each exposure scenario (EPA, 2004). Had a screening-level RQ exceeded an LOC, the exposure scenario would have proceeded to the refined effects determination. When the RQ was less than the LOC, the exposure scenario was not considered further. Additional information on each of these phases of the screening-level effects determination is presented below.

3.13.1.1 Aquatic Exposure Assessment

Exposure of aquatic- (i.e., egg and larval life stages) and terrestrial-phase CRLFs and their prey to aldicarb was estimated using Tier 2 PRZM/EXAMS (Predicted Root Zone Model (PRZM) and Exposure Analysis Modeling System (EXAMS) modeling (see text box description). For each of the exposure scenarios, results were derived for summed peak 1-day (acute exposure) and 21-day average (chronic) concentrations of aldicarb in surface waters of a standard pond (10,000 m^2, 2 m deep). Use of summed peak concentrations for aldicarb and each of the metabolites was a conservative approach for estimating risk to aquatic biota because:

> **Pesticide Root Zone Model (PRZM3)/ EXposure Analysis Modeling System (EXAMS II)**
>
> PRZM predicts pesticide transport and transformation in the soil profile and is capable of simulating pesticide runoff, erosion, plant uptake, leaching, transformation, foliar washoff, and volatilization. Daily edge-of-field loadings of pesticides dissolved in runoff waters and sorbed to sediment, are discharged into a standard water body ("standard pond") simulated by the EXAMS model. Input data are required for soil characteristics, hydrology, pesticide chemistry, meteorology, use pattern, and the crop of interest. For each scenario, PRZM uses 30 years (1961-90) of site-specific climate data to generate exposure results.
>
> EXAMS II takes the runoff and spray drift loadings (see AgDRIFT description below) generated by PRZM and estimates the concentration in the pond on a day-to-day basis.
>
> The simulated results from EXAMS are used to determine a distribution of annual peak concentrations for varying exposure durations (e.g., 24 hour, 96 hour, 21 day, 60 day, one year) based on the 30-year simulation.

- It assumed that aldicarb and the metabolites elicit the same level of effects on aquatic biota. However, aldicarb is more toxic than aldicarb sulfoxide which, in turn, is more toxic than aldicarb sulfone (EPA, 2006).

- The exposure analysis was conducted for a static water body, a conservative approach compared to a moving water body.

- The use of peak 1-day concentrations in the acute assessment was conservative given that the effects data used in the assessment came from studies that were 24 to 96 hours in duration.

3.13.1.2 Terrestrial Exposure Assessment

For direct effects to terrestrial-phase CRLFs and indirect effects to their amphibian prey, the Pacific tree frog, the total daily intake (*TDI*) model was used to estimate dietary exposure to aldicarb. Exposure to other prey items of the CRLF and the Pacific tree frog including fish, and aquatic and terrestrial invertebrates are also discussed below.

The *TDI* exposure model used was:

$$TDI = FIR \bullet FT \bullet \sum_{i=1}^{n} C_i \bullet P_i$$

where
TDI	=	Total daily intake (mg/kg bw/d)
P_i	=	Proportion of ith food item in the diet (unitless)
FIR	=	Normalized food intake rate (kg ww/kg bw/d)
C_i	=	Concentration of pesticide in i^{th} food item (mg/kg ww)
FT	=	Fraction of time in the contaminated area (unitless)

Measured food intake rates (*FIR*) are not available for CRLFs, Pacific tree frogs or other closely related species, primarily due to the difficulties in measuring intake for free-ranging wildlife. Measured food intake rates determined using captive animals should not be used because such animals do not expend energy foraging for food and water, avoiding predators, defending territories, *etc.* (EPA, 1993). Thus, food intake rates estimated for captive animals considerably underestimate expected food intake rates for free-ranging animals. Instead, an allometric equation developed from measurements of free metabolic rate (*FMR*) in free-ranging animals was used to estimate food intake rate for CRLF and Pacific tree frog. Food intake rate is derived from *FMR* using the following equation:

$$FIR \ (g \ / \ day) = \frac{FMR \ (cal \ / \ day)}{\sum_{i=1}^{n} AE_i \ x \ GE_i}$$

where AE_i is the assimilation efficiency of the ith food item (unitless) and GE_i is the gross energy of the ith food item (kcal/g). These data were obtained from EPA (1993).

Allometric equations developed from measurements of free metabolic rate (*FMR*) in free-ranging animals were used to estimate food intake rate for CRLFs and the Pacific tree frog. *FMR* was calculated using the following equation (EPA, 1993):

$$FMR(cal/day) = 0.288 \, (dry \, mass)^{0.878} \, (mg) \quad \text{(Amphibians)}$$

Information on the body weight and dietary preferences of California red-legged frogs and Pacific tree frogs were obtained from Fellers and Guscio (2004), Jameson *et al.* (1970), Hayes and Tennant (1985) and Johnson and Bury (1965).

Terrestrial-phase CRLFs and Pacific tree frogs are unlikely to directly ingest granular, bait and treated seeds because their diet is composed of vertebrate and invertebrate species (Section 3.4). They mainly forage for food along the shoreline and on the surface of the water. Thus, exposure of adult CRLFs and Pacific tree frogs to granular aldicarb near their foraging areas need not be considered in screening-level effects determination. Prey could be exposed to granular aldicarb on treated fields and subsequently move into CRLF foraging areas. For reasons explained in Section 3.6.3, mammals were eliminated as a source of contaminated prey for terrestrial CRLFs. The Fischer and Bowers (1997) nomogram model of pesticide concentrations on invertebrates was used to estimate potential exposure of CRLFs to aldicarb residues on invertebrate prey. The nomogram calculated the maximum and mean residues of aldicarb on these prey items within the treated field immediately after application.

A conservative estimate of the concentration of aldicarb in fish and aquatic invertebrates was calculated using the following equation (Arnot and Gobas, 2004):

$$BCF = \frac{C_{food}}{C_{water}} = (\, Flipid \cdot K_{ow} + OMF_{nl} \cdot OMOPC_{nl} \cdot K_{ow} + Fwater)$$

Where *Flipid* is the lipid fraction (kg/kg) of the species, K_{ow} is the octanol-water partitioning coefficient (unitless), OMF_{nl} is the non-lipid organic matter fraction (i.e., carbohydrate and protein) (kg/kg) of the species, $OMOPC_{nl}$ is the nonlipid organic matter-octanol proportionality constant (i.e., proportionality constant for the partitioning of carbohydrates and proteins) (unitless), and *Fwater* (kg/kg) is the water fraction of the species.

3.13.1.3 Effects Assessment

The screening-level effects determination for aquatic- and terrestrial-phase CRLFs used the most sensitive and relevant measure of effect for the assessment endpoint under consideration. An evaluation of all available data was conducted to ensure that the "best scientific and commercial data available" were used in the effects determination. Toxicity data deemed acceptable by EPA for use in the EFED assessment (EPA, 2006) were accepted for use in the screening-level effects determination. The data presented in

EPA (2006) included data submitted to support the registration process, data compiled by EPA during the review process, and data available from the ECOTOX database. Effects data that were not reviewed by EPA (2006), but were identified in the open literature, were screened using criteria designed to ensure that only high quality data were used in the effects determination. The criteria used to evaluate studies from the open literature are presented in Appendix E.

Table 3-11 lists the effect metrics that were used in the screening-level effects determination. Generally EC/LC_{50} data for acute effects and NOEC data for chronic effects were selected.

3.13.1.4 Risk Characterization

Risk characterization integrates the exposure and effects assessments to determine the potential risk for various exposure scenarios for both direct and indirect effects on the CRLF. In the screening-level effects determination, risks were determined using risk quotients (RQs) that were calculated by dividing estimated environmental concentrations (EECs) by acute and chronic effect metrics (e.g. LC_{50}, NOEC). For acute effects the RQs were compared to the Agency's levels of concern (LOCs) for endangered species (> 0.1 for birds and mammals and > 0.05 for aquatic animals). Chronic risks were evaluated using the chronic LOC > 1. Had one or more of the risk quotients been greater than an LOC that scenario would have continued to a refined effects determination for further assessment.

3.13.2 Uncertainties, Strengths and Limitation of the Assessment

Uncertainties in the problem formulation and assessment of exposure and effects can influence the characterization of risks. The sources of uncertainty in the effects determination were identified, and their magnitude and the direction of their influence were specified. This section concludes with a discussion of the strengths and limitations of the effects determination.

3.13.3 Final Conclusions on Risks of Aldicarb to California Red-Legged Frogs

The information presented in this section summarizes the risk conclusions and effects determination for the CRLF, given the uncertainties, strengths and limitations of the discussed in Section 3.13.3. The information used to derive the effects determination conclusions were based on "best scientific and commercial data available".

The effects determination concluded either "no effect", "may affect, but unlikely to adversely affect" or "likely to adversely affect" for each assessment endpoint (i.e., direct and indirect effects). In general, the exposure scenario(s) assigned the risk category of greatest concern ("no effect" < "may affect, but unlikely to adversely affect" < "likely to adversely affect") for a particular assessment endpoint drove the overall risk conclusion for aldicarb. A determination of "no effect" implies that all exposure scenarios have a RQ<LOC. If one or more of the RQs had been greater than the corresponding LOC those scenarios would have proceeded to a refined effects determination where a risk

conclusion of "may affect, but unlikely to adversely affect" or "likely to adversely affect" would have been made depending on whether the exposure scenarios of greatest concern were categorized as low, intermediate or high risk.

4.0 Screening-Level Effects Determination

The screening-level effects determination used the same risk quotient (RQ) approach as in the EFED assessment to determine if estimated environmental concentrations (EECs) of aldicarb exceeded effects concentrations for California red-legged frogs and their prey. The effects determination was conducted in the three phases – exposure assessment, effects assessment, risk characterization – and is described in the remainder of this section.

4.1 Exposure Assessment

The exposure assessment was carried out separately for species that inhabit aquatic and terrestrial environments. Exposure of aquatic-phase Calfornia red-legged frogs and aquatic prey of terrestrial-phase CRLFs (e.g., fish, aquatic invertebrates) was assessed by estimating concentrations of aldicarb in surface waters (Section 4.1.1). For terrestrial-phase CRLFs and their terrestrial vertebrate prey, exposure to aldicarb was assessed using a total daily intake model (*TDI*) approach (Section 4.1.2).

4.1.1 Aquatic Exposure Assessment

Tier 2 PRZM/EXAMS (Predicted Root Zone Model (PRZM) and Exposure Analysis Modeling System (EXAMS)) modeling were conducted to estimate summed peak concentrations of aldicarb and the carbamate metabolites (aldicarb sulfoxide, aldicarb sulfone) in surface waters of a standard pond (10,000 m^2, 2 m deep) resulting from each of the risk scenarios (Table 3-13). Use of summed peak concentrations for aldicarb and the metabolites is a conservative approach for estimating exposure to aquatic biota because it assumes that aldicarb and the metabolites elicit the same level of effects on aquatic biota. In reality, aldicarb is generally more toxic than aldicarb sulfoxide which, in turn, is more toxic than aldicarb sulfone (EPA, 2006a). PRZM/EXAMS models exposure for a static water body, and thus produce more conservative estimates than would be found in moving water bodies.

For each scenario (Table 3-13), PRZM used 30 years (1961-90) of site-specific climate data to generate exposure results. Results derived using PRZM were used as input into EXAMS. EXAMS simulates the fate, transport, and exposure concentrations of synthetic organic chemicals, including pesticides in limnetic and benthic zones of aquatic systems. The simulated results from EXAMS were used to determine a distribution of annual peak concentrations based on the 30-year simulation. The 1-in-10 year annual peak concentrations for each scenario were compared to the outputs of the effects assessment (Section 4.2) to estimate risk. This too was a conservative approach because peak exposure concentrations are being compared to toxicity effects metrics derived from tests with 24- to 96-hour exposure durations.

The input and output files from the simulation modeling are archived at Bayer CropScience and can be obtained on compact disk (CD) on request.

4.1.1.1 Exposure Assessment Models

PRZM (Version 3.12; June, 1999)

The Pesticide Root Zone Model (PRZM) is a one-dimensional, flow and transport model that can be used to simulate chemical movement in unsaturated soil systems within and immediately below the plant root zone (EPA, 2003). It has two major components - hydrology (and hydraulics) and chemical transport. The hydrologic component simulates the surface and subsurface flow of water. The surface runoff simulation is based on a modified algorithm of the Soil Conservation Service (SCS) curve number technique. The subsurface water movement through the soil compartments (or layers) is simulated by the storage routing technique (also know as the "tipping-bucket method") that utilizes generalized soil parameters such as field capacity, wilting point, saturation water content, and bulk density. The chemical transport component can simulate the movement of pesticides applied on to the soil or on the plant foliage. Biodegradation can also be considered in the root zone. Dissolved, sorbed, and vapor-phase concentrations in the soil are estimated by simultaneously considering the processes of pesticide uptake by plants, surface runoff, erosion, decay, volatilization, foliar wash-off, advection, dispersion, and retardation. PRZM 3.12 also has the ability to simulate as many as three chemicals simultaneously as separate compounds or as a parent-metabolite relationship.

The model has a daily time-step and the outputs can be summarized for a daily, monthly, or annual duration. Runoff, sediment load, dissolved and sorbed chemical loads can be written to sequential files (PRZM-EXAMS transfer files) during program execution, which can be imported into a surface water simulation using EXAMS.

EXAMS (Version 2.98.04; July, 2002)

The Exposure Analysis Modeling System (EXAMS) is an interactive computer program to evaluate the behavior of chemicals in aquatic environments based on general laboratory descriptions of the chemical (EPA, 2003). The model estimates the exposure, fate, and persistence of a compound after its release into the aquatic system. EXAMS uses physically-based relationships to define these three processes. The program consists of a set of unit process models that are founded on the law of conservation of mass.

The EXAMS model treats ionization and partitioning of the compound with sediments and biota discretely for each model segment, thus including the effects of spatial variability of system inputs. The model uses user-defined kinetic rate constants to compute transformation kinetics for processes such as photolysis, hydrolysis, biolysis, and oxidation reactions. The outputs from the model include maximum average Environmental Exposure Concentrations (or EECs) for 24-h, 96-h, 21-d, 60-d, and 90-d time periods, along with maximum instantaneous and mean annual EECs in the system compartments such as water column, sediment, and pore water.

4.1.1.2 Aldicarb Application

TEMIK® 15G brand aldicarb is applied to the soil using methods that, in most cases, almost instantaneously incorporate (cover) the granules with soil after application. Various types and brands of application equipment are used to apply TEMIK® 15G and the application methods can be broadly classified as: 1) in-furrow, shanked or drilled, 2) banded and 3) broadcasted. Brief descriptions of these methods are presented in the following sections. Detailed descriptions of these application methods along with profiles and descriptions of application for each crop are presented in a separate report (Hall *et al.*, 2005). To be conservative, the screening-level effects determination used the application method (i.e., banded) that leaves the greatest number of granules on the soil surface, and thus produce higher modeled exposure concentrations. Banded application is infrequently used for this reason (<4% of aldicarb applications, see Table 3-7).

In-furrow, Shanked or Drilled

With this application method, granules are directed into a thin stream with a width of 1 inch or less. Thin streams of granules are typically directed into open furrows that are immediately covered with 1 inch or more of soil. In addition, concentrated streams may be shanked or drilled into the soil and covered with 2 or more inches of soil using bedding (row forming) equipment. Concentrated streams can also be "side dressed" or shanked approximately 4 inches to the side of plants to a depth of 1 to 4 inches. 92.5% of the applications of TEMIK® 15G take place using this method (Hall *et al.*, 2005, see Table 3-7).

Banded

With this method, granules are distributed into a band that ranges in width from 4 to 12 inches or wider depending on the crop. The method is often referred to as "banded or banding". Banded application may take place through one of the following forms: 1) T-band, 2) in-furrow, 3) light incorporation, and 4) over-the-top post-emergence. Seven and one half percent of the applications of TEMIK® 15G take place using this method (Hall *et al.*, 2005, see Table 3-7).

Incorporation Efficiency

Table 3-7 summarizes the incorporation efficiency expected with these application methods. These are based on field experiments conducted by Bayer CropScience (1988). Along with these field results, the incorporation efficiencies assumed by EPA in the EFED assessment (EPA, 2006a) are also summarized. To be conservative, this exposure assessment used the default EPA incorporation efficiencies which are lower than those observed by Bayer CropScience (1988).

Application Rates

Using a worst-case approach in the screening-level effects determination, exposure modeling was conducted using the maximum application rates for each of the exposure scenarios listed in Table 3-13. Exposure modeling of the cotton and pecan scenarios was conducted using the maximum application rates listed in Table 3-13. The estimated

exposure concentrations (EECs) for the sugar beet exposure scenario were derived by applying a scaling factor to the EECs resulting from the cotton scenario. That is, the ratio of the EECs between maximum application rates for cotton and sugar beets was assumed to be the same as the ratio of the application rates. Because linear sorption is used in PRZM and EXAMS, this assumption is true provided the application dates in both cases are the same.

4.1.1.3 Exposure Assessment Scenarios

A standard surface water exposure assessment scenario consists of a hypothetical treated field of 10 ha, draining into a 1 ha pond that is 2 m deep. As conservative assumptions, drainage in and out of the pond, rainfall over the pond, and evaporation from the pond are not simulated. The only route of dissipation of chemicals reaching the pond is through degradation. Table 4-1 summarizes the exposure assessment scenarios used in the screening-level effects determination. The simulations were conducted for all scenarios using 30 years of daily weather data from an appropriate location (Table 4-1).

Table 4-1. Input assumptions for PRZM/EXAMs modeling by State and crop.

Scenario	MLRA[1]	Weather Station (City, State)	Weather (Years)	Soil Series (Hydrologic Group)
California: Cotton Sugar beets Pecans	17	W93193 (Fresno, CA)	1961 - 90	Twisselman clay (C)

[1]Major Land Resource Area.

The Environmental Fate and Effects Division (EFED) of EPA has developed a graphical user interface (shell), PE4.pl, to conduct simple Tier II surface water exposure assessments (EPA, 2003). When input parameters for PRZM and EXAMS models are entered into PE4.pl, it first executes PRZM (terrestrial medium) and creates a transfer file that serves as input for the EXAMS model. The EXAMS model is then executed by PE4.pl and the EECs in the water column of the water body are extracted from the outputs of the model. PE4.pl is not designed to simulate multiple applications with different application rates.

4.1.1.4 Input Parameters for PRZM and EXAMS

In the case of PRZM and EXAMS, the physical parameters such as the treated field size, geometry of the water body and slope of the field and the hydrological parameters such as runoff Curve Number (CN), erosion parameters, soil parameters, and meteorological data were previously determined by EPA and are built into the standard exposure assessment scenarios (EPA, 2003) used here. In addition, for Tier II exposure assessments, EPA has established a guidance document (EPA, 2002) to derive the chemical parameters used in the models. A refined Tier II assessment was conducted for aldicarb. The derivation of key input parameters is described below. Some of the

parameters used in this modeling effort were originally derived in the EPA Draft RED document (EPA, 2006a).

Partition Coefficients

PRZM and EXAMS consider adsorption and desorption processes through linear isotherms without hysteresis. The adsorption/desorption studies conducted in the laboratory describe the sorption process using non-linear Freundlich isotherms. The Freundlich adsoption coefficient (K_f) and the corresponding organic carbon partition coefficient (K_{oc}) were used in the exposure modeling done for aldicarb.

Adsorption coefficients for aldicarb, aldicarb sulfoxide and aldicarb sulfone were determined in laboratory batch equilibrium studies (Dykes, 1990; Skinner, 1995a,b). The measured adsorption K_{oc} values for aldicarb ranged from 24.7 to 79.2 in four soils, with the K_f values ranging from 0.20 to 0.60. The measured mean K_{oc} value for aldicarb was 49.4. The K_f values for aldicarb sulfoxide and aldicarb sulfone ranged from 0.17 to 0.36 and 0.12 to 0.22, respectively. The corresponding K_{oc} values ranged from 11 to 32 and 13.3 to 74.3 for aldicarb sulfoxide and aldicarb sulfone, respectively. Both batch equilibrium studies for aldicarb sulfoxide and sulfone were conducted on the same set of four soil types and one sediment type. The lowest value K_f value from the above studies (0.12) was used as the linear sorption coefficient (K_d) for the total carbamate residues (TCR) in the PRZM and EXAMS modeling.

Degradation in Soil

Jones and Estes (1995) reviewed studies that characterized the degradation of total carbamate residues (TCR) in the soil (unsaturated zone) and in the saturated zone (ground water). Based on terrestrial field dissipation studies conducted in 24 locations across the United States, the degradation half-life of TCR in soil ranged from 0.3 to 3.5 months (9.1 to 106.5 days). The highest degradation rate from a study conducted in California (soil half-life = 60.83 days) was used with the exposure scenarios.

Degradation in Aquatic Systems

Degradation of aldicarb, aldicarb sulfoxide and aldicarb sulfone in aquatic systems was studied in laboratory sediment-water systems (MRID Nos. 45592107, 45592108, 45592109). The aldicarb aerobic aquatic half-life was 5.5 days. Although the half-life was based on the total system, it represented the water half-life because aldicarb was only detected in the sediment in one interval at low concentrations. The aldicarb sulfoxide aerobic aquatic half-life was 5 days for both the total system and the water phase (from EFED Data Evaluation Record). The aldicarb sulfone aerobic aquatic half-life was 3.5 days for the whole system and water phase (from EFED Data Evaluation Record). Although the data evaluation records note some minor deficiencies with regard to material balances in the sulfoxide and sulfone studies, these deficiencies do not affect the calculation of the half-lives of these compounds.

In an ongoing repeat study at the registrant facility, the preliminary (unaudited) half-life values of aldicarb sulfone were 3.0 and 2.4 days for aerobic and anaerobic aquatic degradation, respectively, with excellent material balances. The rapid aquatic degradation of aldicarb under aerobic conditions is also supported by a guideline study (Skinner, 1995c) submitted to EPA, which showed a half-life of 9 hours. Based on these studies, a half-life value of 5 days was used for the TCR in both dissolved (aerobic) and benthic (anaerobic) phases of the water body.

4.1.1.5 Aquatic Exposure Modeling Results

Summary statistics for the scenarios modeled using PRZM/EXAMS are presented in Table 4-2. The highest EECs were modeled for the application of TEMIK® to sugar beets using two applications of 2.10 lbs ai/A. This scenario used an application start date of January 25 and a 21 day interval between applications. The modeled 1-in-10 year peak 1-day concentration was 0.535 μg/L, the 21-day average concentration was 0.306 μg/L, and the annual average concentration was 0.0239 μg/L. The high surface water concentrations estimated for this scenario are attributed to the high level of precipitation received in California during January when TEMIK® is applied to sugar beets. Most precipitation occurs between November and March at the Fresno weather station used to model the scenarios. Higher levels of precipitation will increase surface run-off and interflow, thus resulting in higher surface water concentrations of aldicarb and the toxic metabolites.

The application of TEMIK® to pecans was modeled at the maximum application rate of 4.95 lbs ai/A with an application start date of April 15. The peak 1-day, 21-day average, and annual average concentrations estimated for this scenario were 0.00929 μg/L, 0.00485 μg/L, and 0.000339 μg/L, respectively.

The lowest EECs were modeled for cotton, the crop the accounts for 99% of the use of aldicarb in California (Cal DPR, 2005). The cotton scenario was modeled with an application start date of May 1 and using applications of 1.05 and 2.10 lbs ai/A separated by a 21 day interval. The peak 1-day, 21-day average, and annual average EECs modeled for cotton were 0.00000258 μg/L, 0.00000122 μg/L, and 0.0000000809 μg/L, respectively. The low estimated surface water concentrations are likely due to the low levels of precipitation that California receives in May.

Table 4-2. Exposure scenarios modeled using PRZM/EXAMS.

Crop	Application Method	Soil Incorporation Efficiency	Application Rate (lbs ai/A)	Peak Concentration	EEC (μg/L) 21-Day Average	Annual Average
Cotton	In-furrow at planting and side dressed	99%	1.05 + 2.10	0.00000258	0.00000122	0.0000000809
Sugar beets	In-furrow at planting and side dressed	99%, 85%	2.10 + 2.10	0.535	0.306	0.0239
Pecans	Band application with light incorporation	85%	4.95	0.00929	0.00485	0.000339

4.1.1.6 Special Routes of Exposure

Subsurface Flow Contribution to Surface Water

Contributions through saturated subsurface flow (shallow groundwater flow) to surface water bodies are important especially during the time period between rainfall events. The exposure assessment presented in this report using PRZM and EXAMS modeling does not account for the subsurface transport of carbamate residues of aldicarb to surface water bodies. The potential importance of this issue is considered here.

Jones *et al.* (1987) reported the results from a surface water and groundwater monitoring study conducted in the vicinity of Lake Hamilton in Florida. Although aldicarb residues were present in the groundwater immediately adjacent to the stream at concentrations up to 171 µg/L, aldicarb residues were not detected in stream or sediment samples, likely due to dilution and degradation of the pesticide.

USGS National Water Quality Assessment (USGS-NAWQA) has an extensive database of monitoring data for aldicarb and its carbamate metabolites, aldicarb sulfoxide and aldicarb sulfone. In almost all of the sampling locations, samples were taken intermittently for extended periods of time. Aldicarb in surface water samples could be the result of overland flow during rainfall events or baseflow (groundwater flow) between rain events. A sampling location where aldicarb or its carbamate metabolites were detected was chosen to analyze overland flow and baseflow contributions of aldicarb to surface waters. Cow Castle creek near Bowman, South Carolina (USGS Station Id: 02174250) was chosen for this analysis. Samples were taken from the creek during the period from April to October of 1996, with a sampling interval ranging from 4 to 17 days. The catchment area is 23.4 sq. miles (approximately 15,000 acres) and the dominant land cover is cropland. The daily stream flow data for this site were obtained from USGS. Using a baseflow separation procedure by Arnold *et al.* (1995) and Arnold and Allen (1999), the baseflow at this site was estimated from the daily stream flow records. The baseflow separation software used in this analysis may be obtained from http://www.brc.tamus.edu/swat/soft_baseflow.html.

Figure 4-1 shows the daily stream flow and estimated baseflow at this site. The carbamate residues of aldicarb measured in the stream at this site clearly correspond to the overland flow periods and the surface water samples taken during baseflow periods had no detections of aldicarb residues.
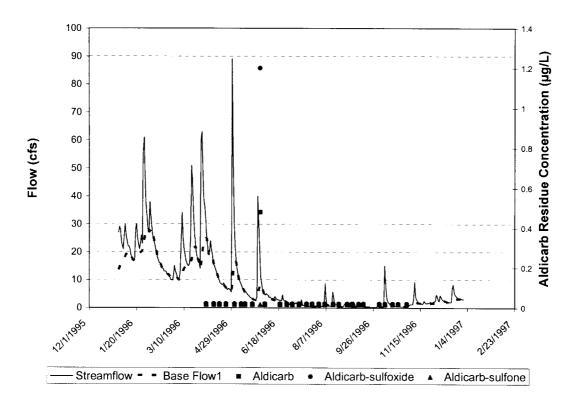
Figure 4-1. **Measured overland flow and estimated baseflow at USGS Station 02174250 along with measured carbamate residues of aldicarb.**

Aldicarb residues in ground water rarely result in aldicarb residues in surface water. One factor limiting movement of aldicarb carbamate residues in ground water into surface water is that the sediment in stream beds is usually highly anaerobic and aldicarb carbamate residues degrade rapidly under anaerobic conditions. Because residues must pass through this sediment bed to enter the surface water, under most circumstances residues are degraded in the sediments before reaching surface water. Another factor that reduces residue levels in groundwater as they move into surface water is the large amount of dilution that occurs as the groundwater is mixed with the water flowing in the stream. The most common mechanism for introduction of aldicarb residues into surface water is interflow to adjacent ditches. Conditions for such a mechanism to be significant are relatively rare and explain the reason for the low frequency of detections of aldicarb and its carbamate metabolites in surface water monitoring programs.

Irrigation Tail Water

The contribution of irrigation tail water from surface irrigation methods (furrow and flood irrigation) to surface water bodies can be a significant. The transport of aldicarb residues from fields irrigated at the surface that are most common in cotton fields in California was not addressed in the exposure assessment using PRZM and EXAMS modeling. The potential importance of this issue is explored here.

Surface water monitoring data from California Department of Pesticide Regulations (CA-DPR) showed no significant detections of aldicarb residues in surface water samples collected between 1992 and 1998. The results are summarized in Table 4-3.

TEMIK® brand aldicarb is incorporated on the ridges of cotton crop at planting, while the irrigation water is applied to the furrows. Due to this separation and also to efficient incorporation, TEMIK® granules are seldom in direct contact with surface irrigation water. Therefore, irrigation tail water is not expected to be a significant source of aldicarb residues transported to surface water.

Table 4-3.   Summary of aldicarb monitoring in surface waters by California DPR.

County	Aldicarb			Aldicarb Sulfoxide			Aldicarb Sulfone		
	No. of Samples	No. of Detects*	Sample Dates	No. of Samples	No. of Detects*	Sample Dates	No. of Samples	No. of Detects*	Sample Dates
Colusa	3	0	2/10/92 to 2/24/92	0	0	-	0	0	-
Contra Costa	1	0	2/10/92	0	0	-	0	0	-
Imperial	113	0	3/15/93 to 2/14/94	65	0	3/15/93 to 1/24/94	65	0	3/15/93 to 1/24/94
Merced	165	0	4/26/91 to 6/12/95	194	0	4/26/91 to 6/12/95	194	0	4/26/91 to 6/12/95
Monterey	65	0	8/1/94 to 8/1/95	65	0	8/1/94 to 8/1/95	65	0	8/1/94 to 8/1/95
Sacramento	63	0	1/20/96 to 8/13/98	53	0	11/26/96 to 8/13/98	53	0	11/26/96 to 8/13/98
San Joaquin	44	0	4/18/91 to 2/10/93	24	0	4/18/91 to 2/10/93	24	0	4/18/91 to 2/10/93
Solano	3	0	2/10/92 to 3/23/92	0	0	-	0	0	-
Sonoma	51	0	8/16/94 to 8/8/95	51	0	8/16/94 to 8/8/95	51	0	8/16/94 to 8/8/95
Stanislaus	409	1 (0.12)	2/25/91 to 10/24/98	438	1 (0.28)	2/25/91 to 12/29/93	438	2 (0.26)	2/25/91 to 12/29/93
Sutter	58	0	1/27/92 to 11/7/94	52	0	11/15/93 to 11/7/94	52	0	11/15/93 to 11/7/94
Yolo	22	0	11/7/96 to 4/15/98	22	0	11/7/96 to 4/15/98	22	0	11/7/96 to 4/15/98

Notes:
*Maximum concentration in µg/L is given in parenthesis.

4.1.2   Terrestrial Exposure Assessment

4.1.2.1   Exposure Assessment Models

The total daily intake (*TDI*) model was used to estimate dietary exposure to aldicarb for direct effects to terrestrial-phase CRLFs and indirect effects to their amphibian prey, the Pacific tree frog. To account for the different diets of juvenile and adult CRLFs, the *TDI* was modeled separately for each of these life stages. The exposure model used was:

$$TDI = FIR \bullet FT \bullet \sum_{i=1}^{n} C_i \bullet P_i$$

where

TDI	=	Total daily intake (mg/kg bw/d)
P_i	=	Proportion of ith food item in the diet (unitless)
FIR	=	Normalized food intake rate (kg ww/kg bw/d)

$$C_i \quad = \quad \text{Concentration of aldicarb in } i^{\text{th}} \text{ food item (mg/kg ww)}$$
$$FT \quad = \quad \text{Fraction of time in the contaminated area (unitless)}$$

Food intake rate was derived from *FMR* using the following equation:

$$FIR\ (g/day) = \frac{FMR\ (cal/day)}{\displaystyle\sum_{i=1}^{n} AE_i \times GE_i}$$

where AE_i is the assimilation efficiency of the ith food item (unitless) and GE_i is the gross energy of the ith food item (kcal/g).

Allometric equations developed from measurements of free metabolic rate (*FMR*) in free-ranging animals were used to estimate food intake rate for CRLFs and their amphibian prey. *FMR* was calculated using the following equation for amphibians (EPA, 1993):

$$FMR\ (cal/day) = 0.288\ (dry\ mass)^{0.878}\ (mg)$$

This *FMR* equation was derived from studies by Feder (1981, 1982) examining oxygen consumption by ranid tadpoles.

4.1.2.2 Input Parameters for Exposure Modeling

Proportion of Dietary Items

California red-legged frogs are opportunistic feeders that prey on virtually anything that is available (Cook, 1997). Hayes and Tennant (1985) reported 196 prey items from 42 taxa in the digestive tract of 31 California red-legged frogs. The most common prey groups were carabid and tenebrionid beetles, water striders, lycosid spiders, and larval neuropterans. The number of prey items identified in the frogs were distributed among the following categories: arachnida (7 of 196 prey items observed), amphipoda (2 of 196), isopoda (22 of 196), insecta (157 of 196), mollusca (6 of 196), fish (undigested dorsal spines observed in one frog), amphibia (2 of 196), and mammalia (1 of 196). The authors noted that prey occurrences in CRLFs reflect prey availability (e.g., current hatch of insects). Larger CRLFs (>8 cm) were observed consuming larger prey (>2 cm). Pacific tree frog (*Hyla regilla*) and California mouse (*Peromyscus californicus*) were the largest prey consumed by CRLFs measuring >10 cm (Hayes and Tennant, 1985). These prey were observed less often, but are considered energetically important representing over half of the prey mass consumed by larger frogs.

The *TDI* model was run separately for juvenile (≤6.5 cm) and large adult (>10 cm) CRLFs to account for the differences in diet between these life stages. Juvenile and adult CRLFs <10 cm in length do not consume vertebrate species (e.g., mice, frogs) as do large

adult CRLFs (>10 cm in length) (Hayes and Tennant, 1985). Exposure to adult CRLFs between 6.5 and 10 cm in length was not modeled. Because CRLFs are opportunistic feeders and because most of their prey have aquatic and terrestrial life stages, it is difficult to determine what proportion of their diet will consist of aquatic versus terrestrial invertebrates. For juvenile CRLFs, the proportion of dietary items was assumed to be 50% aquatic invertebrates and 50% terrestrial invertebrates. Hayes and Tennant (1985) reported that over half of the prey mass of larger CRLFs (>10 cm in length) is composed of vertebrates (e.g., mice and frogs). Taking this into consideration, the diet of adult CRLFs was assumed to consist of amphibians (20%), mammals (20%), fish (20%), aquatic invertebrates (20%), and terrestrial invertebrates (20%). The diets selected for the juvenile and adult CRLF is somewhat arbitrary because of the limited information that exists on dietary composition of the CRLF. However, it will be demonstrated that regardless of proportion of dietary items, the RQ never exceeds the LOC of 0.1 for terrestrial wildlife.

Pacific tree frogs (*Hyla regilla*) primarily forage in shrubs and bushes for terrestrial and flying insects, such as leaf hoppers, midges, and crane flies. They consume lesser quantities of ants, beetles and spiders (Herbert, 2002; Owen, 2000). Johnson and Bury (1965) observed that insects constituted 73.5% of the winter diet of 135 Pacific tree frogs. Their prey includes a number of small invertebrate species that are associated with aquatic and moist habitats (Johnson and Bury, 1965). More adult insects were consumed than larvae, indicating that Pacific tree frogs prefer flying prey. At one of the two study locations the following prey occurrences were observed: collembola (8.2% site 1; 1.4% site 2), lepidoptera larvae (14.8% site 1; 4.1% site 2), hymenoptera (6.6% site 1; 4.1% site 2), diptera adult (14.5% site 1; 8.1% site 2), diptera larvae (8.2% site 1; 1.4% site 2), coleoptera adults (32.8% site 1; 14.9 site 2), coleoptera larvae (19.7% site 1; 4.1% site 2), arachnida (13.2%), plecoptera (2.7% site 2), hemiptera (6.8% site 2), arancae (9.9% site 1; 6.8% site 2), isopoda (9.9% site 1), chilopoda (4.9% site 1), pulmonata eggs (1.6% site 1), pulmonata (11.3% site 1), megadrili (4.9% site 1). These prey items are predominantly found in terrestrial environments. Specifically, 81% and 19% of the tree frog diet consisted of terrestrial and aquatic invertebrates, respectively (Johnson and Bury, 1965). These proportions were used in the initial *TDI* model run for Pacific tree frogs, and were refined in a subsequent tree frog *TDI* model run. Some terrestrial invertebrates that are part of the tree frog diet are not very mobile (e.g., beetles, snails, mites, ticks, worms, etc.). If exposed to aldicarb, these prey items have little chance of being ingested by off-site CRLF because of their low mobility. The proportion of terrestrial invertebrates was separated into mobile and non-mobile prey. The refined proportions were 58%, 23% and 19% for mobile terrestrial invertebrates, non-mobile terrestrial invertebrates and aquatic invertebrates, respectively (Johnson and Bury, 1965).

Concentration of Aldicarb in Prey

Concentrations of aldicarb were required for the following prey items: fish, amphibians, and aquatic and terrestrial invertebrates. The concentration of aldicarb in amphibians was assumed to be the same as fish, based on the assumption that fish and amphibians have similar lipid, non-lipid and water fraction contents. The use of aldicarb

concentration in fish as a surrogate for amphibian concentration was used because methods and data are not available to estimate concentrations of pesticides in amphibians. It was believed that fish was, therefore, the best surrogate for amphibians. The end result of the ESA was not affected by this assumption based on the following. If we assume that the concentration of aldicarb in amphibians was equal to the estimated concentration in terrestrial invertebrates, the highest estimated concentration of aldicarb in CRLF prey, the RQs still do not exceed the LOC of 0.1. The concentration of aldicarb in mammals was zero for reasons provided in Section 3.6.3.

The concentration in fish (and amphibians) and aquatic invertebrates was estimated using the bioconcentration factor (BCF) for aldicarb. A BCF is the ratio of the average concentration of a substance in the tissues of an organism to the average measured concentration of the substance in the water in which the organism occurs.

Generally, substances with a low solubility in water and high octanol-water and octanol-carbon partition coefficients have high BCFs. Aldicarb does not have these properties (Table 3-5). It is highly soluble in water (6,000 mg/L) and has low K_{ow} and K_{oc} values. A low K_{ow} indicates that aldicarb has a low affinity for fatty tissues, and thus is unlikely to biomagnify to higher trophic levels (EPA, 2006a). Based on these properties, several sources have reported that the bioconcentration and biomagnification of aldicarb to higher trophic levels is not a significant exposure pathway (EPA, 2006a; Howard, 1991; Smith, 1992; Johnson and Finley, 1980, WHO, 1991). The bioconcentration of aldicarb is further reduced by the rapid metabolism of the pesticide that occurs when it is ingested. The parent compound and its metabolites are mostly excreted within the first 24 hours of exposure and complete elimination occurs in approximately 5 days (Baron and Merriam, 1988). These properties are typical of carbamate insecticides which elicit rapid nervous system effects that are quickly reversed upon removal of the pesticide (Baron, 1991).

The only measured BCF value found for aldicarb in fish was reported by Metcalf and Sanborn (1975). The BCF value reported was 41.7. These investigators measured the bioconcentration of aldicarb and the metabolites in a model-ecosystem using radio-labelled pesticide. Aldicarb was applied to a 20-gallon aquarium with terrestrial and aquatic media at a dosage of 1.5 mg/aquaria. The aquaria contained salt-marsh caterpillars, sorghum plants, *Daphnia magna*, algae and snails. On day 26 of the 33-day study, 300 mosquito larvae were added. Four days later, 50 of the mosquito larvae were removed for analysis. On day 30, three mosquito fish (*Gambusia affinis*) were added to the aquarium to consume the daphnia and mosquito larvae. The concentration of aldicarb and its metabolites in water, mosquito larvae, and mosquito fish following the 33-day study are provided in Table 4-4.

Table 4-4. **Concentrations of aldicarb in water (mg/L) and aquatic biota following 33-day exposure study (Metcalf and Sanborn, 1975).**

	Water (mg/L)	Proportion of Total	Mosquito larvae	Mosquito fish

Table 4-4. **Concentrations of aldicarb in water (mg/L) and aquatic biota following 33-day exposure study (Metcalf and Sanborn, 1975).**

	Water (mg/L)	Proportion of Total	Mosquito larvae	Mosquito fish
Total ^{14}C	0.16		17.0	2.32
Aldicarb	0.031	20%	16.7	1.31
Unknown metabolite	Trace	-	-	1.01
Aldicarb sulfoxide	0.04	26%	-	-
Aldicarb sulfone	0.056	37%	-	-
Origin	0.025	16%	0.3	-

However, a number of factors call into question the accuracy of this study and BCF value, including:

- The study was conducted over 33 days. However, the half-life of aldicarb in water is 3 to 6 days. Therefore, the concentration that was measured was likely made up largely of the metabolites of aldicarb, most of which are not toxic. The half-lives of the toxic metabolites aldicarb sulfoxide and sulfone in aerobic aquatic environments are 5 and 3.5 days, respectively (MRID No. 45592107). Thus, the concentration that was used to derive the BCF was largely made-up of the non-toxic metabolites of aldicarb.

- Metcalf and Sanborn (1975) noted that the greater the number of degradates a pesticide has, the less likely the parent compound is to bioconcentrate. Thus, the BCF value 41.7 could be considered high given that aldicarb has 18 known metabolites.

- The concentrations reported in Table 4-4 show that aldicarb only contributed about 20% (0.031 mg/L) of the measured concentration of aldicarb and the metabolites. Thus, the BCF is largely based on the concentration of metabolites and not the parent compound.

- The mosquito fish used to derive the BCF were introduced to the test at day 30 and the test was terminated at day 33. They were observed consuming mosquito larvae and *Daphnia magna* that had been in the system for the first 30 days of the test. The mosquito larvae and *D. magna* likely contained some residues of aldicarb and the metabolites, thus the measured BCF accounts not only for the concentrations of aldicarb in water, but also the concentrations in prey.

Based on these confounding factors, the results of Metcalf and Sanborn (1975) were not considered in this effects determination. Instead, the approach used by Arnot and Gobas (2004) was used to estimate the bioconcentration potential of aldicarb in aquatic prey. A conservative estimate of the concentration of aldicarb in fish and aquatic invertebrates was calculated using the following equation (Arnot and Gobas, 2004):

$$BCF = \frac{C_{food}}{C_{water}} = (\ Flipid \cdot K_{ow} + OMF_{nl} \cdot OMOPC_{nl} \cdot K_{ow} + Fwater)$$

where C_{food} is the concentration in food items, C_{water} is the concentration in surface water predicted by PRZM/EXAM, $Flipid$ is the lipid fraction (kg/L) of the species, K_{ow} is the octanol-water partitioning coefficient (unitless), OMF_{nl} is the non-lipid organic matter fraction (i.e., carbohydrate and protein) (kg/L) of the species, $OMOPC_{nl}$ is the nonlipid organic matter-octanol proportionality constant (i.e., proportionality constant for the partitioning of carbohydrates and proteins) (unitless), and $Fwater$ (kg/L) is the water fraction of the species calculated as 1 - (lipid fraction + non-lipid fraction). Values for these input parameters are provided in Table 4-5.

The Arnot and Gobas (2004) approach yielded a BCF of 1.62 for fish and 1.09 for aquatic invertebrates. A BCF of 1.48 for fish was predicted by EPIWIN using BCFWIN v1.15. A Netherlands RIVM report presented a calculated BCF of 1.74 for aldicarb (RIVM, 2000). The latter two BCFs support the results obtained with the Arnot and Gobas (2004) equation. These BCFs are more in line with what would be expected given aldicarbs physical-chemical properties and rapid metabolism of aldicarb in organisms. Among these values, the BCF of 1.62 for fish and 1.09 for aquatic invertebrates were selected for the effects determination because Arnot and Gobas (2004) showed that the majority of model predictions provided good results, within a factor of two of the empirical data, for compounds having a $logK_{ow}$ between 1 and 9 (Arnot and Gobas, 2004). Using this equation, the concentration in prey items were estimated by multiplying the BCF by the one-day peak concentration (acute) in surface water predicted using PRZM/EXAMS model (Table 4-2). The BCF values are presented in Table 4-5.

Table 4-5. **Input parameters and results using Arnot and Gobas (2004) modelling approach.**

Exposure Scenario	Definition	Parameter	Units	Value
Cotton	One-day peak concentration in surface watera	C_{water}	mg/L	0.000446
Sugar beets				0.000595
Pecans				0.000160
Cotton,	Lipid fractionb	$Flipid_{fish}$	kg/kg	0.058
Sugar		$Flipid_{invertebrates}$	kg/kg	0.016
beets and	Octanol-water partitioning coefficientc	K_{ow}	unitless	13.5
Pecans	Non-lipid organic matter fractiond	OMF_{nl}	kg/kg	0.2
	Nonlipid organic matter-octanol proportionality constante	$OMOPC_{nl}$	unitless	0.035
	Water fraction of	$Fwater_{fish}$	kg/kg	0.74

Table 4-5. **Input parameters and results using Arnot and Gobas (2004) modelling approach.**

Exposure Scenario	Definition	Parameter	Units	Value
	the species F	$Fwater_{invertebrates}$	kg/kg	0.78
Calculated Values				
Cotton, Sugar beets and Pecans	Bioconcentration factorg	$BCF_{fish/amphibian}$	L/kg	1.62
		$BCF_{invertebrates}$	L/kg	1.09
Cotton	Concentration in fish/amphibianh	$C_{fish/amphibian}$	mg/kg	0.00072
	Concentration in aquatic invertebratesh	$C_{invertebrates}$	mg/kg	0.00049
Sugar beets	Concentration in fish/amphibianh	$C_{fish/amphibian}$	mg/kg	0.00096
	Concentration in aquatic invertebratesh	$C_{invertebrates}$	mg/kg	0.00065
Pecans	Concentration in fish/amphibianh	$C_{fish/amphibian}$	mg/kg	0.00026
	Concentration in aquatic invertebratesh	$C_{invertebrates}$	mg/kg	0.00017

[a] Calculated using PRZM/EXAMS (Table 4-2).
[b] Averages from data obtained from Morrison et al. (1999), Morrison et al. (1997), Drouillard et al. (1996), Oliver and Niimi (1988) and Russel (1996). A summary of these studies is available in the SETAC Supplemental Data Archive, Item ETC-23-10-002 of Arnot and Gobas (2004).
[c] See Table 3-5.
[d] See Morrison et al. (1997), Russel (1996) and Gobas and Arnot (2004). Value assumed to be the same for fish and aquatic invertebrates.
[e] See Gobas et al. (1999) and Arnot and Gobas (2004).
[f] Calculated using 1 - (lipid fraction + non-lipid fraction).
[g] The biotic constituents were assumed to have a density of 1 kg/L. This is a typical and broadly applied assumption. The calculated BCF was divided by this density to convert from kg/kg to L/kg.
[h] The concentration in prey is calculated by multiplying the BCF by C_{water}.

Nomograms were used to estimate residues of aldicarb on terrestrial invertebrates to assess potential direct effects to the California red-legged frog. Earlier versions of the nomograms for pesticide residues on crops following foliar application were developed by Hoerger and Kenaga (1972). They measured residues of 28 plant protection products on 60 different crops following foliar application and produced nomograms for short grass, long grass, leaves and leafy crops, small seeds and forage crops, pods, cereals, and fruits (Table 4-6). Given the lack of data for residues on terrestrial invertebrates (insects), Kenaga (1973) proposed that the nomograms for forage crops and cereals be used for small and large insects, respectively. These values are inappropriate for the assessment of aldicarb because they do not account for granular pesticides.

Table 4-6.   Typical and maximum residue values (normalized for an application rate of 1 kg a.i./ha) (Hoerger and Kenaga, 1972; Kenaga, 1973).

Plant/Plant part	Typical Value	Maximum Value
Short grass	112	214
Long grass	82	98
Leaves and leafy crops	31	112
Small seeds/Forage crops/Insects	29	52
Pods	2.7	11
Cereals/Large insects	2.7	8.9
Fruit	1.3	6.3

Recent studies have focused on improving the nomograms for insects and have considered both foliar and soil-incorporated granular applications (Fischer and Bowers, 1997; Brewer *et al.*, 1997; Joermann, 1998). Fischer and Bowers (1997) compiled measured data from field studies conducted in the late 1980s through the early 1990s. Residue values were collected within 24 hours of 175 foliar applications and 56 soil applications (liquid and granular formulations). Twenty-two of the soil applications used granular formulations, and thus were applicable to the assessment of aldicarb. The nomogram values based on these 22 sites are presented in Table 4-7. The complete data set used to calculate the nomograms is presented Appendix F. The sites used to derive the nomograms in Table 4-7 included applications to potatoes, sweet potatoes, corn, and citrus. The application of aldicarb to citrus crops is not permitted in California. However, these data were retained to account for the application of aldicarb to pecans. The nomogram values derived by Fischer and Bowers (1997) for insects at sites that applied pesticides using soil-incorporation are lower than the values for foliar application (Table 4-6). This is because incorporating pesticides into the soil greatly reduces the potential for contact with insects.

Table 4-7.   Summary statistics of pesticide residues in terrestrial invertebrates, normalized to mg/kg/1 lb a.i./A (Fischer and Bowers, 1997).

Application method	Mean	Standard Deviation	Geometric Mean	Median	Maximum	Minimum
Soil incorporated	0.211	0.295	0.0846	0.156	1.28	0.006

The nomograms in Table 4-7 were used to estimate the concentration of aldicarb in terrestrial invertebrates (Table 4-8). The results are presented using the mean and maximum nomogram values in Table 4-8.

Table 4-8.   Summary of exposure scenarios, estimated environmental concentrations of aldicarb and residues predicted on terrestrial invertebrates using Fischer and Bowers (1997) nomograms.

Scenario / Appl. Rate (lbs ai/A)	Nomogram Values for Terrestrial Invertebrates		Terrestrial Invertebrate Concentrations (mg/kg ww)[*]	
	Mean	Maximum	Based on Mean	Based on Maximum
Cotton	0.211	1.28	0.66	4.03

Table 4-8.   Summary of exposure scenarios, estimated environmental concentrations of aldicarb and residues predicted on terrestrial invertebrates using Fischer and Bowers (1997) nomograms.

Scenario / Appl. Rate (lbs ai/A)	Nomogram Values for Terrestrial Invertebrates		Terrestrial Invertebrate Concentrations (mg/kg ww)*	
	Mean	Maximum	Based on Mean	Based on Maximum
1.05 + 2.10 lbs ai/A				
Sugar beets			0.89	5.38
2.10 + 2.10 lbs ai/A				
Pecans			1.04	6.34
4.95 lbs ai/A				

*The terrestrial vertebrate concentration (mg/kg ww) is derived by multiplying the nomogram value by the application rate (lbs ai/A).

Fraction of Time in the Contaminated Area

Bulger *et al.* (2003) reported that CRLFs spend most of their time within 5 meters of their summer aquatic habitat. During rain events they may move as far as 130 meters to upland areas where they remain for approximately 4 to 6 days. However, 90% of the individuals observed remained within 60 meters of water at all times. Aldicarb is applied directly to fields and not to the aquatic environment. Using a very conservative approach, the screening-level effects determination assumed that CRLFs spend 100% of their time in areas containing aldicarb residues.

Gross Energy (GE) (kcal/g wet weight)

Gross energy values are reported in the *Wildlife Exposure Factors Handbook* (EPA, 1993). A mean *GE* value of 0.88 kcal/g was used in the TDI model for aquatic invertebrates. This value was derived from the *GE*s for bivalves (0.80 kcal/g), isopods and amphipods (1.1 kcal/g), and cladocerans and insect larvae (0.74 kcal/g) (Cummins and Wuycheck, 1971; Golley, 1961; Tyler, 1973; Jorgensen *et al.*, 1991; Pierotti and Annett, 1987; Minnich, 1982; Thayer *et al.*, 1973). The mean *GE* of 1.6 kcal/g used for fish was derived from the values for bony fishes (1.2 kcal/g) and small fish (2.0 kcal/g) (Cummins and Wuycheck, 1971; Thayer *et al.*, 1973; Erickson and Elsner, 1981; Miller, 1978). The *GE*s for grasshoppers and crickets (1.7 kcal/g) and beetles (1.5 kcal/g) were averaged to derive the *GE* value 1.6 kcal/g for terrestrial invertebrates (Collopy, 1975; Bell, 1990). The *GE*s for mice and frogs are 1.7 kcal/g and 1.2 kcal/g, respectively (Gorecki, 1975; Golley, 1960; Koplin *et al.*, 1980).

Assimilation Efficiency (AE) (unitless)

The EPA Exposure Factors Handbook (EPA, 1998) reports *AE*s for the prey of birds and mammals but not amphibians. In lieu of *AE*s for the prey of amphibians, the *AE*s reported for the prey of birds were used as a surrogate for the prey of CRLFs. The *AE*s for the prey of birds are lower than the values for mammalian prey, and thus more conservative. Adult CRLFs consume fish and aquatic invertebrates and terrestrial vertebrates and invertebrates. The *AE*s of prey from these categories are reported in Table 4-11. Pacific tree frogs and juvenile CRLFs consume aquatic and terrestrial

invertebrates. The *AE*s for these prey items are presented in Tables 4-11 and 4-12 for juvenile CRLFs and Pacific tree frogs, respectively (EPA, 2003).

Body Weight

Body weight data for the California red-legged frog were available from a ten-year (1991-2000) study of threatened invertebrates from four streams in San Luis Obispo County, California (Scott and Rathbun, 2001). Body length and weight data were collected for 459 California red-legged frogs. Body lengths ranged from 3.5 to 13.9 cm and weights ranged from 4.3 to 247 g. Juvenile and large adult CRLFs were considered to measure $\leq$ 6.5 cm and $\geq$ 10 cm in length, respectively (Hayes and Tennant, 1985). The weight of juveniles ranged from 4.3 to 27 g with an average weight of 12.2 g (=72). The weight of adults ranged from 74 to 247 g with an average weight of 131.9 g (=387). The raw data for these calculations are presented in Appendix G.

To account for the different diets of juvenile and adult CRLFs, the *TDI* was modeled separately for each of these life stages. The *FMR* equation uses the dry mass of the species of interest. Thus, the juvenile and large adult body weights were converted to dry weights. EPA (1993) reports that amphibians are 85% water. Therefore, dry weight = wet weight × (1 - 0.85), or 1.8 g for juvenile CRLFs and 19.8 g for adults. Feder (1981) provides an equation for calculating the dry mass of ranid tadpoles of 0.047 (wet mass)$^{1.06}$ (mg). This method produces lower dry mass values that in turn produce lower *FMR*s and result in lower estimates of total daily intake. The former approach was considered more appropriate for this assessment because it results in more conservative predictions of intake.

Adult Pacific tree frogs range in length from 2.5 to 5 cm (USGS, 2003). Jameson *et al.* (1970) sampled 10 Pacific tree frogs from each of nine locations in British Columbia, Oregon, California and Baja California. The body lengths and weights of the sampled frogs are presented in Table 4-9. They reported a grand mean weight of 2.27 g. Using the equation to convert to dry weight produces a value of 0.341 g.

Table 4-9.   Pacific tree frog body lengths and weights reported in Jameson *et al.* (1970).

Location	Group Mean Weight (g)	Snout-Vent Length (cm)	Shank Length (cm)
British Columbia			
Chiliwack	1.60	2.96	1.52
Oregon			
Lee's Camp	4.45	3.87	2.09
Madras	3.22	3.32	1.74
California			
Crescent Mills	1.79	2.81	1.58
San Diego	2.38	3.34	1.68
Willows	1.68	2.93	1.47
Julian	2.22	3.32	1.61

Table 4-9. **Pacific tree frog body lengths and weights reported in Jameson *et al.* (1970).**

Location	Group Mean Weight (g)	Snout-Vent Length (cm)	Shank Length (cm)
Baja California			
Calamajue	1.62	2.79	1.36
Palmarito	1.47	2.70	1.43
Grand Mean	**2.27**		

In summary, the input parameters described above and used in the *TDI* models for adult California red-legged frogs, juvenile CRLFs, and Pacific tree frogs are presented in Tables 4-10, 4-11, and 4-12, respectively.

Table 4-10. **Input variables used to model total daily intake of aldicarb by adult California red-legged frogs.**

Variable	Parameters		
Body weight (*BW*; g)	0 = 131.9		
Free metabolic rate (*FMR*; cal/day)	$Y=aBW^b$ (a=0.288, b=0.878)		
Gross energy (*GE*; kcal/g ww)			
Fish	1.6		
Aquatic invertebrates	0.88		
Terrestrial invertebrates	1.6		
Mammals	1.7		
Amphibians	1.2		
Assimilation efficiency (*AE*; unitless)			
Fish	0.79		
Aquatic invertebrates	0.77		
Terrestrial invertebrates	0.72		
Mammals	0.78		
Amphibians	0.79		
Proportion of Prey Items in Diet			
Fish	0.20		
Aquatic invertebrates	0.20		
Terrestrial invertebrates	0.20		
Mammals	0.20		
Amphibians	0.20		
Concentrations in Food Items	Cotton	Sugar beets	Pecans
Fish (mg/kg ww)	0.0000000417[a]	0.000865[a]	0.0000150[a]
Aquatic invertebrates (mg/kg ww)	0.00000000282[a]	0.000586[a]	0.0000102[a]
Terrestrial invertebrates (mg/kg ww)	4.03[b]	5.38[b]	6.34[b]
Mammals (mg/kg ww)	0[c]	0[c]	0[c]
Amphibians (mg/kg ww)	0.0000000417[a]	0.000865[a]	0.0000150[a]
Fraction of Time in Contaminated Area	1		

[a] Used most conservative peak concentration predicted by PRZM/EXAMS to estimate concentration in biota.

[b] Used most conservative (i.e., maximum) estimated concentration in terrestrial invertebrates (Table 4-8).

[c] Mammals in the diet of CRLFs are not expected to contain residues of aldicarb (see Section 3.6.3 for further discussion).

Table 4-11. **Input variables used to model total daily intake of aldicarb by juvenile California red-legged frogs.**

Variable	Parameters		
Body weight (*BW*; g)	0 = 12.2		
Free metabolic rate (*FMR*; cal/day)	$Y=aBW^b$ (a=0.288, b=0.878)		
Gross energy (*GE*; kcal/g ww)			
Aquatic invertebrates	0.88		
Terrestrial invertebrates	1.6		
Assimilation efficiency (*AE*; unitless)			
Aquatic invertebrates	0.77		
Terrestrial invertebrates	0.72		
Proportion of Prey Items in Diet			
Aquatic invertebrates	0.50		
Terrestrial invertebrates	0.50		
Concentrations in Food Items	Cotton	Sugar beets	Pecans
Aquatic invertebrates (mg/kg ww)	0.00000000282[a]	0.000586[a]	0.0000102[a]
Terrestrial invertebrates (mg/kg ww)	4.03[b]	5.38[b]	6.34[b]
Fraction of Time in Contaminated Area	1		

[a] Used most conservative peak concentration predicted by PRZM/EXAMS to estimate concentration in biota.
[b] Used most conservative (i.e., maximum) estimated concentration in terrestrial invertebrates (Table 4-8).

Table 4-12. **Input variables used to model total daily intake of aldicarb by Pacific tree frogs.**

Variable	Parameters		
Body weight (*BW*; g)	0 = 2.27		
Free metabolic rate (*FMR*; cal/day)	$Y=aBW^b$ (a=0.288, b=0.878)		
Gross energy (*GE*; kcal/g ww)			
Aquatic invertebrates	0.88		
Terrestrial invertebrates	1.6		
Assimilation efficiency (*AE*; unitless)			
Aquatic invertebrates	0.77		
Terrestrial invertebrates	0.72		
Proportion of Prey Items in Diet			
Aquatic invertebrates	0.19		
Terrestrial invertebrates	0.81		
Refined Proportion of Prey Items in Diet			
Aquatic invertebrates	0.19		
Terrestrial invertebrates (mobile)	0.58		
Terrestrial invertebrates (non-mobile)	0.23		
Concentrations in Food Items	Cotton	Sugar beets	Pecans
Aquatic invertebrates (mg/kg ww)	0.00000000282[a]	0.000586[a]	0.0000102[a]
Terrestrial invertebrates (mg/kg ww)	4.03[b]	5.38[b]	6.34[b]
Fraction of Time in Contaminated Area	1		

[a] Used most conservative peak concentration predicted by PRZM/EXAMS to estimate concentration in biota.
[b] Used most conservative (i.e., maximum) estimated concentration in terrestrial invertebrates (Table 4-8).

4.1.2.3 Terrestrial Exposure Modeling Results

The *TDI* modeling results are presented in Table 4-13.

Table 4-13. **Total Daily Intake (TDI) results for direct and indirect effects of aldicarb to California red-legged frog and the Pacific tree frog.**

Exposure Scenario	Species and Life Stage	TDI (mg/kg bw/day)
Cotton (1.05 + 2.10 lbs ai/A)	Adult CRLF	0.00905
	Juvenile CRLF	0.0302
	Pacific tree frog	0.0603
	Pacific tree frog – Refined diet[1]	0.0433
Sugar beets (2.10 + 2.10 lbs ai/A)	Adult CRLF	0.0121
	Juvenile CRLF	0.0403
	Pacific tree frog	0.0804
	Pacific tree frog – Refined diet[1]	0.0577
Pecans (4.95 lbs ai/A)	Adult CRLF	0.0142
	Juvenile CRLF	0.0475
	Pacific tree frog	0.0948
	Pacific tree frog – Refined diet[1]	0.0680

[1]The proportions of prey items in the diet of the Pacific tree frog were adjusted to account for the low mobility of prey potentially exposed to aldicarb on treated fields (see Section 4.3.2.2).

4.2 Effects Assessment

Aldicarb inhibits the enzyme acetylcholinesterase (AChE), resulting in the buildup of neurotransmitter acetylcholine (ACh) at cholinergic nerve endings, and causing continual nerve stimulation that can result in death (Risher *et al.*, 1987; WHO, 1991). Although the toxic effects of aldicarb are associated with the inhibition of AChE activities in the central nervous system (CNS), depressed AChE activities may also occur in the plasma, muscle, and brain (Risher *et al.*, 1987; Perkins and Schlenk, 2000). In aquatic animals, symptoms of exposure to toxic levels of aldicarb include hyperactivity, muscular contractions, erratic swimming, altered pigmentation, lethargy, and moribund behavior (Geiger *et al.*, 1990; Gallo *et al.*, 1995; Odin-Feurtet, 1999). In terrestrial animals, symptoms of exposure to toxic levels of aldicarb include perspiration, salivation, muscular weakness, nausea, vomiting, evacuation of bowel and bladder, laboured respiration, ataxia, muscle spasms and convulsions. The effects of aldicarb on AChE are rapidly reversible, and spontaneous recovery from sublethal doses occurs within a matter of hours (Baron and Merriam, 1988). The most significant biological effect of aldicarb and its sulfoxide and sulfone metabolites is acute toxicity resulting from the inhibition of AChE. Baron and Merriam (1988) have shown that repeated exposure to aldicarb is no different than the effects of acute exposure, essentially resembling a string of acute exposure and recovery situations.

The screening-level effects determination for aquatic- and terrestrial-phase CRLFs used the most sensitive and relevant measure of effect for the assessment endpoint under consideration (Table 4-14). An evaluation of all available data was conducted to ensure that the "best scientific and commercial data available" were used in the effects

determination. Toxicity data deemed acceptable by EPA for use in the EFED assessment (EPA, 2006a) were accepted for use in the screening-level effects determination unless more appropriate values were available in the open literature. Studies that were identified in the open literature were screened using criteria designed to ensure that only high quality data were used in the effects determination. The criteria used to evaluate studies from the open literature are presented in Appendix E. A detailed review of the data for aquatic biota and birds (used as a surrogate for terrestrial CRLF) can be found in Appendix D.

Table 4-15 presents the values that were used as the toxicity component of the RQ calculation. These endpoints were selected to meet the measures of effect listed in Table 3-11. For the most part, the effects values used in the EFED assessment were also used in this screening-level effects determination. The exceptions are discussed in the following paragraphs. The EFED assessment can be referenced for a description of each of the studies (EPA, 2006a).

An early life stage NOEC value of 78 µg/L was reported for fathead minnow (*Pimephales promelas*) (Pickering and Gilliam, 1982). EPA did not use this value in the EFED assessment because the most sensitive 96h LC50 value of 52 µg/L for bluegill sunfish (*Lepomis macrochirus*) was lower that the 48h LC50 value of 8,860 µg/L for fathead minnow. Instead, the Agency calculated an ACR of 114 from the acute and chronic toxicity values for fathead minnow. Several acute and chronic toxicity values are available for fathead minnows. EPA chose the values that would create the largest ACR (114), and thus, the lowest estimated no effect concentration (ENEC) of 0.48 µg/L for bluegill sunfish. Such a high ACR (114) is unexpected given the toxicity and mode of action of aldicarb. This insecticide is a fast-acting cholinesterase (ChE) inhibitor with a short aquatic half-life and effects that are reversible upon removal of aldicarb. Similar levels of toxicity are reported for acute and chronic exposure to aldicarb, thus, acute effects are likely to supersede chronic effects. Therefore, the screening-level effects determination used the chronic NOEC of 78 µg/L that was reported for fathead minnow.

The EFED assessment used the mysid shrimp (*Mysidopsis bahia*) NOEC of 1 µg/L as the chronic endpoint for aquatic invertebrates. Rather than using this marine species to represent freshwater invertebrates, the screening-level effects determination used the 21-day NOEC of 18 µg/L for *Daphnia magna* reported in Handley *et al.* (1991).

A chronic toxicity value for birds (used as a surrogate for terrestrial-phase CRLF) was not available for the EFED assessment (see Section 3.11.1). Thus, chronic effects to terrestrial-phase CRLFs were not considered in the effects determination.

Table 4-14. Measures of ecological effect selected for screening-level effects determination.

Species	Exposure Duration	Endpoint	Concentration	Reference
Aquatic Lifestage – Direct Effects				

Table 4-14. **Measures of ecological effect selected for screening-level effects determination.**

Species	Exposure Duration	Endpoint	Concentration	Reference
Bluegill sunfish	Acute	LC50	52 µg/L	Mayer and Ellersieck, 1986
Fathead minnow	Chronic	NOEC	78 µg/L	Pickering and Gilliam, 1982
Terrestrial Lifestage – Direct Effects				
Mallard	Acute	LD50	1 mg/kg	MRID #107398
Avian	Chronic	NA	NA	
Terrestrial Lifestage – Indirect Effects				
Chironomus tentans	Acute	EC50	20 µg/L	Moore *et al.*, 1998
Bluegill sunfish	Acute	LC50	52 µg/L	Mayer and Ellersieck, 1986
Mallard	Acute	LD50	1 mg/kg	MRID #107398
Honey bee	Acute	LD50	0.285 µg/bee	MRID # 00036935
Daphnia magna	Chronic	NOEC	18 µg/L	Handley *et al.*, 1991
Fathead minnow	Chronic	NOEC	78 µg/L	Pickering and Gilliam, 1982

NA: Not available, no chronic studies reported in EPA (2006a).

4.3 Risk Characterization

Risk characterization integrates the exposure and effects assessments to determine the potential risk for various exposure scenarios for both direct and indirect effects on the CRLF. The screening-level effects determination evaluated risks using risk quotients (RQs) that were calculated by dividing estimated environmental concentrations (EECs) by acute and chronic effect metrics (e.g., LC_{50}, NOEAC). For acute effects, the RQs were compared to the Agency's levels of concern (LOCs) for endangered species (>0.1 for birds and mammals and >0.05 for aquatic animals). Chronic risks were determined using the chronic LOC of >1.

4.3.1 Aquatic-Phase California Red-Legged Frogs

The RQs for direct effects to aquatic-phase CRLFs are presented in Table 4-15. The results show that the aquatic acute endangered species and chronic LOCs are not exceeded for any of the exposure scenarios. The highest acute RQ of 0.0103 for aldicarb applied to sugar beets is more than 4 times lower than the acute endangered species LOC of 0.05.

Table 4-15. **Risk quotients for direct effects (acute and chronic) to aquatic-phase California red-legged frogs.**

Exposure Scenario	Peak EEC (µg/L)	21-day Average (µg/L)	LC50 (µg/L)	NOEC (µg/L)	Acute RQ (EEC/LC50)	Chronic RQ (EEC/NOEC)
Cotton (1.05 + 2.10 lbs ai/A)	0.00000258	0.00000122	52	78	0.0000000496	0.0000000156
Sugar beets (2.10 + 2.10 lbs ai/A)	0.535	0.306	52	78	0.0103	0.00392
Pecans	0.00929	0.000485	52	78	0.000179	0.00000622

Table 4-15. Risk quotients for direct effects (acute and chronic) to aquatic-phase California red-legged frogs.

Exposure Scenario	Peak EEC (µg/L)	21-day Average (µg/L)	LC50 (µg/L)	NOEC (µg/L)	Acute RQ (EEC/LC50)	Chronic RQ (EEC/NOEC)
(4.95 lbs ai/A)						

4.3.2 Terrestrial-Phase California Red-Legged Frogs

Direct effects to terrestrial-phase California red-legged frogs were assessed using RQs derived for juvenile and adult CRLFs. Potential indirect effects to the prey of CRLFs were determined by deriving RQs for aquatic vertebrates and invertebrates and terrestrial vertebrates. For aquatic prey, PRZM/EXAMS EECs were used as the exposure component of the RQs. For terrestrial prey, the exposure component of the RQs was determined using the *TDI* approach for vertebrates (i.e., Pacific tree frog). Chronic RQs could not be derived for terrestrial vertebrates (e.g., CRLF, Pacific tree frog) because appropriate chronic effects data were not available for this group. The mode of action of aldicarb and available toxicity data indicate that acute and chronic toxicity levels are similar (see Section 3.8.3). Thus, the RQs derived for acute exposure are expected to account for potential chronic exposures.

4.3.2.1 Direct Effects

None of the RQs for direct effects to juvenile and adult CRLFs exceeded the acute endangered species LOC of 0.1 (Table 4-16). The *TDI* for juvenile and adult CRLFs was derived using the most conservative concentrations of aldicarb in prey items. Peak 1-day concentrations estimated from PRZM/EXAMS and the maximum concentrations estimated from Fischer and Bowers (1997) were used to estimate concentrations of aldicarb in fish, amphibians, and aquatic and terrestrial invertebrates.

For juvenile and adult CRLFs, determining the proportion of prey items was a somewhat arbitrary process because of the limited information that exists on diet composition of the CRLFs. However, regardless of proportion of dietary items, the RQ never exceeded the acute endangered species LOC. For example, worst case "what if" dietary scenarios were investigated for juvenile and adult CRLF using application of aldicarb to pecans. Exposure of aldicarb to pecans was selected because the application rate for aldicarb is highest for this crop, and produced the highest RQs. The worst case scenarios assumed that 100% of the diet of juvenile and adult CRLFs was terrestrial invertebrates. This assumption is based on the estimated concentration of aldicarb in terrestrial invertebrates that is six orders of magnitude greater than concentrations estimated in aquatic biota. The risk quotients assuming 100% terrestrial invertebrates in the diet of juvenile and adult CRLFs were 0.0950 and 0.0711, respectively, which are below the corresponding LOC of 0.1.

The concentration of aldicarb in Pacific tree frogs, a prey item of adult CRLFs, was assumed to be equal to that of fish. Assuming that the concentration of aldicarb in

Pacific tree frogs was equal to the estimated concentration in terrestrial invertebrates, the highest estimated concentration of aldicarb in CRLF prey, the RQ would have been 0.0315. Thus, this most conservative scenario still showed no risk to adult CRLF.

Another way to estimate a conservative concentration of aldicarb in mobile terrestrial insects is to consider the acute dermal honey bee toxicity test reported in the EFED assessment (MRID # 00036935). Because aldicarb is an insecticide, there must be a limit to how high the residue load in an insect can be before it can no longer move from a treated field to the habitat of the CRLF. If an insect received a lethal or near lethal dose of aldicarb, it would become immobilized and incapable of moving offsite. The honey bee acute dermal LD50 value of 0.285 µg/bee could be used as a surrogate species to estimate the dose or body burden that would immobilize an insect (MRID # 00036935). Honey bees weigh approximately 81 to 140 mg (Winston, 1987). If one assumes a honey bee weight of 100 mg, the LD50 is 2.85 mg/kg bw. This value is considered a reasonable upper limit of the concentration of aldicarb in mobile terrestrial insect prey of CRLFs. The lowest residue data on terrestrial insects derived using the Fischer and Bowers (1997) approach was 4.03 mg/kg for cotton. This implies that the predictions using the Fischer and Bowers (1997) approach are very conservative and perhaps estimate body burdens on terrestrial insects that would render them immobile and, therefore, unavailable for consumption by CRLFs located well away from agricultural fields.

4.3.2.2 Indirect Effects

Aquatic Prey

Table 4-16 resents the RQs for indirect effects to CRLF aquatic prey. None of the RQs for aquatic vertebrates exceeded the acute endangered species or chronic LOCs for aquatic animals. The RQs for aquatic invertebrates were also below the acute endangered species and chronic LOCs for all of the exposure scenarios. Based on these RQs, individual and population level effects to the aquatic prey of terrestrial-phase CRLFs are not expected from worst-case exposures to total aldicarb residues.

Table 4-16. **Risk qotients for indirect effects (acute and chronic) to aquatic prey of terrestrial-phase California red-legged frogs.**

Exposure Scenario	Peak EEC (µg/L)	1 in 10 year 21-day EEC (µg/L)	LC50 (µg/L)	NOEC (µg/L)	Acute RQ (EEC/LC50)	Chronic RQ (EEC/NOEC)
Aquatic vertebrates						
Cotton (1.05 + 2.10 lbs ai/A)	0.00000258	0.00000122	52	78	0.0000000496	0.0000000156
Sugar beets (2.10 + 2.10 lbs ai/A)	0.535	0.306	52	78	0.0103	0.00392
Pecans (4.95 lbs ai/A)	0.00929	0.000485	52	78	0.000179	0.00000622
Aquatic invertebrates						
Cotton	0.00000258	0.00000122	20	18	0.000000129	0.0000000678

Table 4-16. **Risk qotients for indirect effects (acute and chronic) to aquatic prey of terrestrial-phase California red-legged frogs.**

Exposure Scenario	Peak EEC (µg/L)	1 in 10 year 21-day EEC (µg/L)	LC50 (µg/L)	NOEC (µg/L)	Acute RQ (EEC/LC50)	Chronic RQ (EEC/NOEC)
(1.05 + 2.10 lbs ai/A)						
Sugar beets (2.10 + 2.10 lbs ai/A)	0.535	0.306	20	18	0.0268	0.0170
Pecans (4.95 lbs ai/A)	0.00929	0.000485	20	18	0.000465	0.0000269

Terrestrial Prey

None of the RQs for indirect effects to the Pacific tree frog exceeded the acute endangered species LOC of 0.1 (Table 4-16). The RQ for the Pacific tree frog using the pecan exposure scenario (RQ = 0.0948) was close, however, to the LOC of 0.1. To further demonstrate that Pacific tree frog are not at risk, the *TDI* model was refined by developing more realistic "what if" scenarios on proportion prey items consumed by the species. The diet of the Pacific tree frog primarily consists of terrestrial invertebrates. Specifically, 81% and 19% of the Pacific tree frogs diet consists of terrestrial and aquatic invertebrates, respectively (Johnson and Bury, 1965). Using these dietary proportions in the TDI model resulted in a daily intake of 0.0948 mg/kg bw/day, a value that produced a RQ of 0.0948. Some terrestrial invertebrates that are part of the Pacific tree frogs diet are not very mobile. If exposed to aldicarb, these prey items have little chance of being ingested by off-site CRLFs due to their low mobility. The proportion of terrestrial invertebrates was separated into mobile and non-mobile prey. The refined proportions were 58%, 23% and 19% for mobile terrestrial invertebrates, non-mobile terrestrial invertebrates and aquatic invertebrates, respectively (Johnson and Bury, 1965). Using these refined dietary proportions in the *TDI* model produced a RQ equal to 0.0680 for the pecan exposure scenario. This was repeated for the cotton and sugar beet scenarios and resulted in a 30% reduction in RQ values (Table 4-17).

Table 4-17. **Risk quotients for direct effects and indirect effects to terrestrial-phase California red-legged frogs and Pacific tree frogs.**

Exposure Scenario	Species	TDI (mg/kg bw/day)	LD50 (mg/kg bw/day)	Acute RQ (TDI/LD50)
Cotton (3.15 lbs ai/A)	Adult CRLF	0.00905	1	0.00905
	Juvenile CRLF	0.0302		0.0302
	Pacific tree frog	0.0603		0.0603
	Pacific tree frog – Refined proportion diet	0.0433		0.0433
Sugar beets (4.2 lbs ai/A)	Adult CRLF	0.0121	1	0.0121
	Juvenile CRLF	0.0403		0.0403
	Pacific tree frog	0.0804		0.0804
	Pacific tree frog – Refined proportion diet	0.0577		0.0577
Pecans	Adult CRLF	0.0142	1	0.0142

Table 4-17. **Risk quotients for direct effects and indirect effects to terrestrial-phase California red-legged frogs and Pacific tree frogs.**

Exposure Scenario	Species	TDI (mg/kg bw/day)	LD50 (mg/kg bw/day)	Acute RQ (TDI/LD50)
(4.95 lbs ai/A)	Juvenile CRLF	0.0475		0.0475
	Pacific tree frog	0.0948		0.0948
	Pacific tree frog – Refined proportion diet	0.0680		0.0680

4.4 Sources of Uncertainty, Strengths and Limitations of the Assessment

The screening-level effects determination for California red-legged frogs contains uncertainties. Uncertainties in the problem formulation and assessment of exposure and effects can influence the characterization of risks. It is therefore important to identify the sources of uncertainty in the assessment, and specify the magnitude and direction of their influence. The following sources of uncertainty were identified in the screening-level effects determination:

Problem Formulation

- The conceptual model was intended to define the linkages between the application of aldicarb and potential exposure pathways to aquatic and terrestrial animals that could potentially affect ecological receptors. As such, the conceptual model provides the scientific basis for selecting assessment and measurement endpoints to support the risk assessment process. Potential uncertainties arise from lack of knowledge regarding ecosystem functions, failure to adequately address spatial and temporal variability in the evaluations of sources, fate, and effects, omission of stressors, and overlooking secondary effects (EPA, 1998). The physical and chemical properties, modes of transport, exposure routes, and other characteristics of aldicarb have been extensively researched and documented. Thus, the relationships identified in the conceptual model carry a high level of confidence and are considered a low source of uncertainty in the screening-level effects determination.

Exposure Assessment

- A modeling approach was used to estimate aldicarb concentrations in a hypothetical pond located adjacent to a treated field. Tier 2 PRZM/EXAMS modeling was conducted to estimate summed peak concentrations of aldicarb and the carbamate metabolites (aldicarb sulfoxide, aldicarb sulfone). For each risk scenario, surface water concentrations were predicted for a 10,000 m^2, 2 m deep pond, using 30 years of site-specific climatic data. This spatial scale is considered a reasonable, though conservative, example of the agricultural situations in which aldicarb is used. The summed peak concentrations of aldicarb and the metabolites is a conservative approach for estimating risk to aquatic biota, because it assumes

that aldicarb and the metabolites elicit the same level of effects on aquatic biota. However, the metabolites are generally less toxic than the parent compound. Without appropriate surface water monitoring data to compare with modeled results, the magnitude of this source of uncertainty is unknown, but is likely biased in a conservative direction.

- The PRZM/EXAMS modelling assumed that TEMIK® was used at the maximum application rates and applied using methods that would result in the highest estimated environmental concentrations (e.g., band). In practice, typical application rates are more commonly used, as are application methods that provide higher incorporation of the granules into the soil. Thus, the assumptions that were made in the PRZM/EXAMS modelling biased the effects determination in a conservative direction.

- For acute exposure, the screening-level effects determination derived risk quotients using the peak 1-day 1-in-10 year EEC. Using peak concentrations was a conservative approach given that the effects data used in the assessment came from studies that were 24 to 96 hours in duration. This is considered an important source of uncertainty with a bias towards overestimating risks.

- The screening-level effects determination focused on the direct contact and dietary ingestion routes of exposure. Exposure to aldicarb in sediment and pore water was not considered because aldicarb is not expected to occur at elevated concentrations in sediment. Ingestion of TEMIK® granules by aquatic animals was not considered as an exposure pathway because: 1) aldicarb is highly water soluble and does not transport to aquatic systems as granules, and 2) it is applied directly to the field and not the aquatic environment. Based on this information, the source of uncertainty related to identification of exposure pathways is considered low.

- Drinking water could not be included in the exposure modelling for terrestrial-phase CRLFs and their prey due to the difficulty of quantifying intake. It is unclear how much uncertainty omitting this route of transport contributed to the effects determination.

- Concentrations of aldicarb in aquatic prey of terrestrial-phase CRLFs were not available. Aldicarb residues in prey were predicted using estimated bioconcentration factors. The approach by Arnot and Gobas (2004) was used to estimate a conservative concentration in aquatic prey. The approach yielded a BCF of 1.62 for fish and 1.09 for aquatic invertebrates. Percent lipid content in fish and aquatic invertebrates used to estimate these BCFs were obtained from measured lipid contents for fish and aquatic invertebrates in the Great Lakes. Thus, it was assumed that lipid content in fish and aquatic invertebrates in the Great Lakes were similar to contents found in prey in California. Non-specific non-lipid content values were used to estimate these BCF values because of lack

of information on these parameters. Although the estimated BCF value of 1.62 for fish was supported by two other sources, BCFWIN v1.15 and a Netherlands RIVM report, there is still some uncertainty associated with this BCF because of the factors noted above. Some uncertainty also exists for the BCF value of 1.09 for aquatic invertebrates. The BCFs values of 1.62 and 1.09 for fish and aquatic invertebrates do not take into account the metabolism of aldicarb. These BCF values were then multiplied by conservative surface water concentrations estimated by PRZM/EXAM modeling to derive concentrations of aldicarb in prey. Based on this information, the estimated concentrations in prey likely overestimate risk to CRLFs and Pacific tree frog.

- Concentrations of aldicarb in terrestrial invertebrate prey of terrestrial-phase CRLFs were not available. Aldicarb residues in prey were predicted using the Fischer and Bowers (1997) nomograms. Although uncertainty is introduced in estimates of prey concentration when using these nomograms, this uncertainty likely overestimates risk to CRLFs and Pacific tree frog. It was demonstrated (Section 4.3.2.1) that if the LD50 for honey bees, used as a surrogate for terrestrial invertebrate prey, was 2.85 mg/kg. This body burden would render a honey bee immobile, and consequently terrestrial invertebrates would not be capable of moving offsite. The value of 2.85 mg/kg is lower than any of the estimated terrestrial invertebrate prey concentrations used in the *TDI* modelling. This implies that the predictions using the Fischer and Bowers (1997) approach are very conservative and perhaps estimate body burdens on terrestrial insects that would render them immobile and, therefore, unavailable for consumption by CRLFs inhabiting shorelines and aquatic environements. Thus, the terrestrial invertebrate prey concentrations used in the TDI model likely overestimated the residues of aldicarb on this prey item.

- No measurements of free metabolic rate were available for the California red-legged frog or Pacific tree frog. Similarly, measured food intake rates were not available for California red-legged frog, Pacific tree frog, or a reasonable surrogate species. Therefore, free metabolic rates were estimated using allometric equations. The use of allometric equations introduces some uncertainty into the exposure estimates because they are based on species different from the CRLF and Pacific tree frog. For CRLF and Pacific tree frog, the effects determination used the *FMR* equation for ranid frogs reported in the EPA *Exposure Factors Handbook* (EPA, 1993). California red-legged frogs are from the *Rana* genus, thus, decreasing the uncertainty in using the equation. Pacific tree frogs are from the genus *Hyla*, so using the *FMR* based on *Rana* species leads to uncertainty in the derived *FMR*.

- Diets, particularly for juvenile and adult CRLFs, were derived using limited data. The diet for the juvenile CRLFs was 50% aquatic invertebrates and 50% terrestrial invertebrates. The adult CRLF was assumed to consume 20% each for aquatic invertebrates, fish, amphibians, terrestrial invertebrates and mammals.

These assumptions were evaluated using "what if" scenarios that maximized the risk to the CRLF by assuming that the diet of the CRLF consisted of 100% terrestrial invertebrates, a prey having the highest concentration of aldicarb. These scenarios did not influence the risk conclusions results which remained below the LOC of 0.1. Thus, uncertainties regarding diet have no influence on the risk conclusions of no risk.

Effects Assessment

- There were uncertainties in the effects assessment associated with using toxicity data for fish as a surrogate for aquatic-phase amphibians and data for birds as a surrogate for terrestrial-phase amphibians. When assessing effects to terrestrial-phase amphibians, EPA guidance states that avian toxicity data can be used when data for amphibians are unavailable (EPA, 2004b). However, a comparison of acute LD50s for standard avian test species (mallard and quail) and bullfrogs (*Rana catesbeiana*) showed that bullfrogs are less sensitive (Tucker and Crabtree, 1970). Quite often, bullfrog LD50s were greater than the highest tested dose. The bullfrog is considered a reasonable surrogate for the California red-legged frog because both are large *Rana* species. Thus, the use of avian acute toxicity data as surrogate toxicity data for amphibians is likely to bias the effects determination towards overestimating risks.

4.5 Conclusions

The risk quotients derived in the screening-level effects determination indicate that aquatic-phase California red-legged frogs, terrestrial-phase California red-legged frogs and their prey are not at risk from exposure to aldicarb or the metabolites aldicarb sulfoxide and aldicarb sulfone from the application of TEMIK® according to the label-permitted uses for California. The screening-level effects determination was based on worst-case scenarios and conservative assumptions that produced the highest potential exposures to aldicarb. The conservative RQs and the "what if" scenarios indicate that the juvenile and adult CRLFs and the Pacific tree frogs are not at risk from exposure to aldicarb.

Thus, all of the exposure scenarios that were assessed in the screening-level effects determination were categorized as having "no effect" on aquatic California red-legged frogs, terrestrial-phase California red-legged frogs and their prey.

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Appendix A – Home Ranges of Adult California Red-Legged Frog Terrestrial Prey

California red-legged frogs consume a variety of insect and invertebrate species (Hayes and Tennant, 1985). They have also been reported consuming larger prey such as fish (*Gasterosteius aculeatus*), amphibians (*Hyla regilla*), and mammals (*Peromyscus californicus*) (Hayes and Tennant, 1985). Of these prey, species from the genus *Peromyscus* (e.g., California mouse, deer mouse) and *Hyla* may inhabit areas where aldicarb is applied (e.g., agricultural fields, forests). Although unlikely, there is a potential for CRLFs to consume *Peromyscus* or *Hyla* that were exposed at the site of aldicarb application and then traveled to CRLF habitat. Home range information for *Hyla regilla* species indicates the home range is approximately 33 meters for resident frogs, while movements of up to 400 m have been noted in migrating frogs (Morey, 2005). The home range of *Peromyscus californicus* varies greatly from 150 to 3,788 m^2, with average ranges from 1,161 to 1,500 m^2 (Ribble and Salvioni, 1990; USC, 2006). Home ranges from the common *Peromyscus* species deer mice (*P. maniculatus*) also vary greatly from 242 to 3,000 m^2 (Bunker, 2001). Mean home ranges reported in the EPA *Wildlife Exposure Factors Handbook* for deer mice vary from 140 to 1280 m^2 (EPA, 1993). Using a conservative approach to select from the available data, the maximum home range of 3,788 m^2 for *P. californicus* was used. This value was converted to a radius to estimate the distance from a treated field that a mouse would potentially travel:

$$Area\ of\ a\ circle = \pi R^2$$
$$3,788\,m^2 = \pi R^2$$
$$R = \sqrt{\frac{3,788\,m^2}{\pi}}$$
$$R = 34.7 \approx 35\,m$$

where R = radius (m). Thus, a conservative home range radius of 35 m for California mouse was derived.

Appendix B – Common Weed Seeds of the United States

A comparison of common weed seed characteristics to aldicarb granules was carried out to evaluate the possibility of birds ingesting aldicarb granules because they mistake them for seeds. The shape, size, color, and distribution of seeds from nearly 200 weed species found in agricultural areas of the United States are presented in Table B-1. Data were collected from the U.S. Department of Agriculture's "Common Weeds of the United States" (USDA, 1971). Aldicarb granules are round, black and shiny, and typically range in size from 0.4 to 1.1 mm (Best and Gionfriddo, 1994; Best, L. personal communication, 2005; Uceda and Le Gren, 2001a,b). The majority of species characterized in Table B-1 have seeds that are larger than aldicarb granules and colored differently. Most weed species seeds are ovoid or obovate compared to the round aldicarb granules. It is unclear if these subtle differences in shape would affect bird preference for aldicarb granules. In addition, birds exhibit a lack of preference for the gypsum or graphite coating of aldicarb granules (Best and Gionfriddo, 1994; Best *et al.*, 1996).

Of the nearly 200 weed species seeds described in USDA (1971), approximately seven are similar to aldicarb granules in both color and size. *Portulaca oleracea* seeds are black with a whitish scar on one end and a wrinkled texture. They measure approximately 0.5 mm in length and are found in cultivated areas throughout the United States. This weed is a pest in vegetable, ornamental, citrus, and cotton crops. *Amaranthus retroflexus* and *Spergula arvensis* seeds measure 1-1.2 and 1-1.5 mm in length, respectively. Both weed species are found in cultivated fields and waste areas. Because these species are found in agricultural areas, there is a possibility that birds could mistake aldicarb granules for seeds of these species. Although somewhat similar in size and color to aldicarb granules, *Amaranthus albus*, *Brassica kaber*, *Chrysanthemum leucanthemum*, *Galinsoga parviflora*, and *Halogeton glomeratus* seeds are not associated with agricultural environments or the crops that aldicarb is applied to. These weed species are most commonly found in waste and fallow areas. Therefore, birds are unlikely to be exposed to aldicarb granules in areas where these weed species are present.

Seeds of several weed species are similar in size to aldicarb granules (0.4-1.1 mm), but other factors mitigate against birds mistaking aldicarb granules for seeds in agricultural environments. Weed species such as *Cerastium* spp., *Linaria dalmatica*, *Potentilla norvegica*, and *Verbascum* spp. are found in wastes, abandoned fields, and pasture lands where aldicarb granules are not applied. *Capsella bursa-pastoris, Cuscuta approximata, Descurainia pinnata, Lychnis alba,* and *Stellaria media* are found in agricultural environments and are similar in shape to aldicarb granules. The colors of seeds from these weed species vary in color, but none are black. Thus, birds are unlikely to mistake these seeds for shiny, black aldicarb granules.

Many weed species have black seeds, but which are larger than aldicarb granules. *Agrostemma githago, Chenopodium album*, and *Polygonum* spp. are found in agricultural

environments. Seeds from these weeds are black, and measure 1.3-1.5 mm for *Chenopodium album* and up to 2.5-3 mm for *Polygonum* spp. Birds are unlikely to mistake smaller aldicarb granules for seeds of these weed species. *Agrostemma githago* seeds are triangular in shape. This further decreases the chance that birds will mistake round aldicarb granules for seeds of this species.

Based on the seed characteristics summarized in Table B-1, birds are unlikely to mistake aldicarb granules for common weed seeds. Only three weed species – *Portulaca oleracea, Amaranthus retroflexus* and *Spergula arvensis* – are found in agricultural environments and have shape, size and color characteristics similar to aldicarb granules. Other weed species seeds are similar in size or color to aldicarb, but not both. The potential for confusion is further lessened by the habitat preferences (e.g., meadows, pasture, woodlands) and distributions of weed species in the United States.

Table B-1. Comparison of common weed seed characteristics and distribution in the United States to aldicarb granules.

Scientific Name	Common Name	Shape	Color	Length (mm)	Distribution in United States
Granular aldicarb		Round	Black, shiny	0.4-1.1	NA
Abutilon theophrasti	Velvetleaf, butterprint, piemarker	Kidney-shaped, flat	Grayish-brown	3	All but large area in north central US and southern most area of southern States
Achillea millefolium	Common yarrow	Oblong	White, gray	0.2	All but southwestern US and southern TX
Aegilops cylindrica	Jointed goatgrass	Elongated	NR	8	South central and southwestern US
Agropyron repens	Quackgrass, couchgrass	Elongated	NR	6-22	All but southern US
Agrostemma githago	Corn cockle	Triangular	Black	2-3	All of US
Alternanthera philoxeroides	Alligator weed	Ovoid, heart shaped	NR	1.6	Extreme southeast US, isolated area in CA
Amaranthus albus	Tumble pigweed, tumbleweed	Round	Black, shiny	0.7-1	All of US
Amaranthus blitoides	Prostrate pigweed	Round, flat	Black, shiny	1.4-1.7	All but extreme southern coastal areas and southwest
Amaranthus retroflexus	Redroot pigweed	Oval	Black, dark red-brown, shiny	1-1.2	All of US
Ambrosia artemisiifolia	Common ragweed	Round	Light-brown	4-5	All but northern MN, WI, MI, and MN
Ambrosia psilostachya	Western ragweed	Ovoid	NR	3	Western half of US, except extreme northwest, across northeastern border, except MN
Andropogon virginicus	Broomsedge	Elongated	Brown	3	Eastern half of US, California
Anthemis cotula	Mayweed	Oblong	Brown	0.1	All of US
Apocynum cannabinum	Hemp dogbane, Indian hemp	Elongated	NR	4-6	All of US
Arctium minus	Common burdock	Oblong	Dark-gray	3-7	All but southern border States, northern WI, MN, and MI
Arctium vulgaris	Mugwort	Oblong	NR	0.2	Northeastern quarter of US, Pacific coast States
Asclepias syriaca	Common milkweed	Oval, flat	Brown	6	Eastern half of US, except southern most States
Asclepias verticillata	Eastern whorled milkweed	Oval, flat	Brown	5	Eastern and central US
Astragalus mollissimus	Woolly loco	Oblong	NR	0.2	South central and part of southwestern US

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Report 201692

Table B-1.   Comparison of common weed seed characteristics and distribution in the United States to aldicarb granules.

Scientific Name	Common Name	Shape	Color	Length (mm)	Distribution in United States
Granular aldicarb		**Round**	**Black, shiny**	**0.4-1.1**	**NA**
Avena fatua	Wild oat	Elongated	White, yellow, brown, gray, or black	8.5-10	Across US, except some Atlantic coast, Gulf coast and southern Plains States
Barbaea vulgaris	Yellow rocket	Oblong	Light-yellow, yellowish brown	1-1.5	Eastern US, eastern WA and OR, distinct area in north central US
Bidens bipinnata	Spanish-needles	Elongated	Black, dark-brown	10-18	Eastern half of US, except northern border States
Bidens frondosa	Devils beggarsticks	Oblong	Dark-brown, black	5-10	All of US
Brassica kaber	Wild mustard	Round	Black, purplish-brown	1-1.5	All of US
Brassica nigra	Black mustard	Round	Reddish-brown, black	1.5-2	All but north central US
Bromus commutatus	Hairy chess	Elongated	NR	3.8	Throughout US, less common in southeast, southwest, north central
Bromus japonicus	Japanese brome, Japanese chess	Elongated	NR	4	All US but northern Maine and southern Florida
Bromus secalinus	Cheat, chess	Elongated	NR	4	All but north central US
Bromus tectorum	Downy brome, downy chess, Cheatgrass	Elongated	NR	0.9	All but extreme southeastern US
Brunnichia cirrhosa	Redvine, buckwheat-vine	Elongated	NR	7-10	Extreme southeast, Gulf Coast States
Camelina microcarpa	Smallseed falseflax	Oblong	Dark-brown	<1	Central and northern US
Campis radicans	Trumpetcreeper, Virginia creeper	Winged	NR	15	Eastern half of US, except extreme northern States
Cannabis sativa	Hemp, marijuana	Ovoid	Mottled-brown	3	All but extreme southeast, southwest, and north central US
Capsella bursa-pastoris	Shepherdspurse	Oblong	Yellow, orangish-brown	0.8-1	All of US
Cardaria draba	Hoary cress	Obovate	Reddish-brown	2	All but southern boundary of south central and western US

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Report 201692

Table B-1. Comparison of common weed seed characteristics and distribution in the United States to aldicarb granules.

Scientific Name	Common Name	Shape	Color	Length (mm)	Distribution in United States
Granular aldicarb		**Round**	**Black, shiny**	**0.4-1.1**	**NA**
Carex lasiocarpa	Bull sedge	Ovoid	NR	1.1	All but southeastern US
Carex nebraskensis	Nebraska sedge	Ovoid	NR	1.2	Western half of US, except Pacific coast
Cenchrus echniatus	Southern sandbur	Ovoid	NR	1.6-3.2	Southern coasts and southern border
Cenchrus incertus	Field sandbur, burgrass	Ovoid	Straw, mauve or purple	2.2	Southern US
Cenchrus longispinus	Longspine sandbur	Ovoid	NR	2.2-3.8	Central US, Atlantic coast, isolated western areas
Centaurea diffusa	Diffuse knapweed	Oblong	Dark-brown	2.5	Distinct northwestern area, isolated areas across central US
Centaurea maculosa	Spotted knapweed	Oblong	Brownish	2	Northeastern quarter of US, Pacific coastal States
Centaurea repens	Russian knapweed	Ovoid	Grayish	2-3	Central and western US
Centaurea solstitialis	Yellow starthistle	Oblong	Light, dark-brown	2-3	West coast States, distinct areas across US
Cerastium arvense	Field chickweed	Ovoid	Reddish-brown, chestnut-brown	0.6-1	All but lower southern half of US
Cerastium vulgatum	Mouseear chickweed	Round	Chestnut-brown	0.8	Throughout most of US
Chenopodium album	Common lambsquarters	Round	Black, shiny	1.3-1.5	All of US
Chrysanthemum leucanthemum	Field oxeye-daisy	Obovate	Black	1-1.5	All but north central US
Chrysothamnus nauseosus	Rubber rabbitbrush	Elongated, hairy	NR	5	Western US, except coastal areas
Chrysothamnus viscidiflorus	Douglas rabbitbrush	Elongated, hairy	NR	3	Western US, except coastal areas
Cichorium intybus	Chicory	Obovate	Light-brown	2-3	All but southern and north central US
Cirsium arvense	Canada thistle	Oblong	Light to dark-brown	2.5-3.5	Northern half of US
Cirsium vulgare	Bull thistle	Oblong	Straw-colored	<4	All of US

Table B-1.   Comparison of common weed seed characteristics and distribution in the United States to aldicarb granules.

Scientific Name	Common Name	Shape	Color	Length (mm)	Distribution in United States
Granular aldicarb		**Round**	**Black, shiny**	**0.4-1.1**	**NA**
Conium maculatum	Poison hemlock	Ovoid	Pale-brown, grayish-brown ribbed	2.4	All but north central US
Conringia orientalis	Haresear mustard	Oblong	Dark grayish-brown	2-2.5	All but extreme southeastern and southwestern US
Convolvulus arvensis	Field bindweed	Ovoid	Dark brown-gray	3-5	All but southern TX, NM, AZ and extreme southeast US
Convolvulus sepium	Hedge bindweed	Ovoid	Slate, black	4-5	Central and eastern US, extreme northwestern US
Conyza canadensis	Horseweed	Oblong	NR	1	All of US
Croton capitatus	Woolly croton	Round, ventrally flat	Grayish	3.6	South central and southeastern US
Cuscuta approximata	Smallseed dodder	Oval, oblong	Light-brown	1	Western US, central WA to eastern MT, as far south as northern NM
Cuscuta indecora	Largeseed dodder	Kidney shaped	NR	1-1.7	All but northeastern US
Cuscuta pentagona	Field dodder	Round, flat	Yellow to reddish-brown	1-1.5	All but western coastal, north central and northeastern States
Cynodon dactylon	Bermuda grass, devil grass	Elongated	NR	2	Southern two thirds of US
Cyperus esculentus	Yellow nutsedge	Ovoid	Yellow	1.2-1.5	All but most north central US
Cyperus rotundsu	Purple nutsedge	Oblong	Olive-gray to brown	1.5	South Atlantic and Gulf Coast States, coastal CA and eastern AZ
Datura stramonium	Jimsonweed	Kidney shaped	Dark-brown, black	3.3	All but northwestern and north central US and south TX
Daubentonia punicea	Coffeeweed	Round, oblong	NR	5.5	South Atlantic and Gulf Coast States
Daucus carota	Wild carrot, Queen-Annes-Lace	Oblong, flat, spikes	Light grayish-brown	2-4	All but north central US
Delphinium barbeyi	Tall larkspur	Rectangular	Smoky-brown	2.3	Two distinct areas in western US
Delphinium menziesii	Menzies larkspur	Oblong	Brownish	1.5	North Pacific coast, distinct area in western WA, eastern ID

Bayer CropScience
Report 201692

Table B-1. Comparison of common weed seed characteristics and distribution in the United States to aldicarb granules.

Scientific Name	Common Name	Shape	Color	Length (mm)	Distribution in United States
Granular aldicarb		Round	Black, shiny	0.4-1.1	NA
Delphinium occidentale	Duncecap larkspur	Square	NR	1.3	Distinct area in northwestern US
Delphinium geyeri	Geyer larkspur, plains larkspur	Long, flat	NR	2-3.5	Distinct area in western US
Descurainia pinnata	Tansymustard	Oblong, flat	NR	<1	All of US
Digitaria ischaemum	Smooth crabgrass	Ovoid	NR	1.7	All but southern Florida and southern States near Mexico
Digitaria sanguinalis	Large crabgrass	Ovoid	Black	2	All but far north central US
Diodia teres	Poorjoe	Oval	Light-brown	3-4	Eastern half of US, except extreme northern States
Dipsacus sylvestris	Teasel	Oblong	Grayish-brown	2-3	Northwestern coastal States, band across central US into northeastern US
Echinochloa crus-galli	Barnyard grass	Ovoid	Tan to brown	2.5-3.5	All but extreme southeast US
Eleusine indica	Goose grass, yard grass, silver crab grass, wire grass	Oval	Reddish brown	1-1.5	All but northern Maine, parts of north central and northwestern US
Erigeron annuus	Annual fleabane	Obovate	Straw-colored	1	Eastern half of US, except extreme southeastern States, Pacific coast
Eupatorium capillifolium	Dog fennel	Elongated, hairy	NR	1.3-1.9	Southeastern US
Eupatorium rugosum	White snakeroot	Elongated, hairy	Black, dark-brown	2-2.5	Eastern half of US, except extreme southeastern States
Euphorbia corollata	Flowering spurge	Ovoid	Gray, light-brown	3.5-4.5	Eastern half of US, except northern most States
Euphorbia esula	Leafy spurge	Oval	Light-gray, yellow-brown	2	Northern US
Euphorbia maculata	Spotted spurge, nodding spurge	Oblong	Dark-brown, black	1.1-1.6	Eastern half of US, coastal WA, OR and northern CA
Euphorbia supina	Prostrate spurge	Ovoid	Gray-brown, reddish	0.8-1	Eastern half of US, Pacific coast, two distinct areas of ID and AZ

Page 103

Bayer CropScience
Report 201692

Table B-1. Comparison of common weed seed characteristics and distribution in the United States to aldicarb granules.

Scientific Name	Common Name	Shape	Color	Length (mm)	Distribution in United States
Granular aldicarb		Round	Black, shiny	0.4-1.1	NA
Franseria discolor	Skeletonleaf bursage, bur-ragweed, silverleaf, poverty weed	NR	NR	4-6	Central and southwest US
Galeopsis tetrahit	Hempnettle	Ovoid	NR	3-4	Northeastern quarter of US, eastern WA to western MT
Galinsoga parviflora	Smallflower galinsoga	Elongated	Dark-brown, black	1-5	All but northern border States
Galium aparine	Catchweed bedstraw	Round, pitted, spines	Gray-brown	2-3	All of US
Galium mollugo	Smooth bedstraw	Kidney shaped	NR	1.3	Extreme northeastern US, coasts of OR and CA
Glechoma hederacea	Ground ivy	Ovoid	Dark-brown	1.5-2	Central and eastern US, except extreme southern States
Gutierrezia dracunculoides	Common broomweed	Oval	NR	2	South central US
Halogeton glomeratus	Halogeton, barilla	Round, flat	Brown, black	1	Distinct areas in West central States
Haplopappus tenuisectus	Burroweed	Oblong	NR	0.1	Southern NM and AZ, west TX
Helenium amarum	Bitter sneezeweed	Wedge-shaped	Reddish-brown	1-1.5	Eastern half of US, except northern States
Helianthus annuus	Sunflower	Ovoid	White, gray, dark-brown	4-8	All of US
Holcus mollis	German velvetgrass	Elongated	NR	3	Coasts of MA, PA, NJ, WA, OR, CA
Hordeum jubatum	Foxtail barley	Elongated	Yellow	3	All but southern Atlantic and Gulf Coast States
Hordeum pusillum	Little barley	Elongated	Yellow	3-8	Most of US
Hymenoxys odorata	Bitter rubberweed	Oblong	Grayish	1.6-2.2	Southwestern, south central US
Hypericum perforatum	St. Johns wort	Oblong	Black, shiny	1.5	Eastern half of US and northwest States
Ipomoea hederacea	Ivyleaf morning glory	Round, flat	Dark-brown, black	6	Throughout eastern, southeastern, central, and southwestern US
Iponmoea purpurea	Tall morning glory	Ovoid, angled	Brown, black	4-5	Central and eastern US, Western coastal States

Bayer CropScience
Report 201692

Table B-1.   Comparison of common weed seed characteristics and distribution in the United States to aldicarb granules.

Scientific Name	Common Name	Shape	Color	Length (mm)	Distribution in United States
Granular aldicarb		**Round**	**Black, shiny**	**0.4-1.1**	**NA**
Iva xanthifolia	Marsh elder	Triangular	Gray to black	3	Northeastern US, area from WA to MN and south to NM and TX
Jussiaea decurrens	Winged water primrose	Kidney shaped	NR	0.3-0.4	Eastern half of US, except northern most States
Jussiaea repens	Creeping water primrose	Oblong	NR	1	Gulf Coast States and slightly north, southern NJ
Kochia scoparia	Kochia	Ovoid	Brown, yellow	1.8	Northern half of US, except western States
Lactuca pulchella	Blue lettuce	Elongated, hairy	Dark red-brown	2-3	Area from central WA to Great Lakes and south to Mexican border
Lactuca serriola	Prickly lettuce	Elongated, flat	Pale-brown, gray	3-3.5	All of US
Lamium amplexicaule	Henbit	Oblong-obovate	Grayish-brown	1.5-2.4	All but north central US
Lepidium campestre	Field pepperweed	Ovoid	Dark-brown	2-2.5	Eastern half of US, except southeast, three distinct areas in western US
Lepidium virginicum	Virginia pepperweed	Obovate	Chestnut-brown	1.5-2	All but north central US, southern CA, AZ, NM
Linaria dalmatica	Dalmatian toadflax	Wing-angled	NR	0.8-1	Distinct areas in western, central, and northeastern US
Linaria vulgaris	Yellow toadflax, butter-and-eggs	Round, flat	Dark-brown, black	1.5-2	All of US
Lithospermum arvense	Corn gromwell	Ovoid	Grayish-tan	3	All but eastern MT to north central MN
Lonicera japonica	Japanese honeysuckle	Ovoid, flat	Dark-brown, grayish-brown	0.3	Southeastern quarter of US
Lychnis alba	White cockle	Round, oval	Pale-gray	1	Northern half of US
Mava neglecta	Common mallow	Ovoid, flat	Reddish-brown	1	All of US
Medicago lupulina	Black medic	Oval	Yellowish-green, greenish-brown, orange-brown	1.5-2	All of US
Mollugo verticillata	Carpetweed	Kidney shaped	Orange-red	1.2-1.5	All but north central US

Table B-1. Comparison of common weed seed characteristics and distribution in the United States to aldicarb granules.

Scientific Name	Common Name	Shape	Color	Length (mm)	Distribution in United States
Granular aldicarb		Round	Black, shiny	0.4-1.1	NA
Muhlenbergia schreberi	Nimblewill	Elongated	Reddish-brown	1-1.4	Most of central and eastern US
Orobanche minor	Clover broomrape	Ovoid	NR	0.2	Atlantic coast from NJ to NC
Oxalis stricta	Common yellow woodsorrel	Flat	Brown	1-1.3	All of US
Oxytropis lambertii	Lambert crazyweed	Kidney shaped	Brown, black	1.5	Central US
Panicum dichotomiflorum	Fall panicum	Oval	NR	2	All but parts of north central US and TX
Paspalum dilatatum	Dallis grass	Round	NR	2	Southern Atlantic, Pacific, and Gulf Coasts
Paspalum distichum	Knotgrass	Oval	NR	2.2-2.8	Mideastern, southern, and western US
Phalarisk arundinacea	Reed canary grass	Elongated	NR	3-4.2	All but eastern CA, south Atlantic and Gulf Coast States
Physalis heterophylla	Clammy ground cherry	Obovate	Light-orange, straw	2	Eastern half of US
Phytolacca americana	Pokeweed	Round, ovoid	Black, shiny	3	Eastern half of US
Plantago lanceolata	Buckhorn plantain	Ellipsoid	Brown, black, shiny	2-3	All of US
Plantago major	Broadleaf plantain	Angled	Light to dark-brown	1-1.7	All of US
Plantago rugelii	Blackseed plantain	Oval	Dark-brown	1.5-2.5	Eastern half of US
Polygonum amphibium	Water smartweed	Ovoid	NR	2.5-3	Northern half of US, except western States
Polygonum aviculare	Prostrate knotweed	Ovoid	Dark reddish-brown, black, dull to lustrous	2-2.5	All of US
Polygonum coccineum	Swamp smartweed	Oval	Dark brown, black, shiny	2.5-3	All but south Atlantic and Gulf Coast States, and east central US
Polygonum convolvulus	Wild buckwheat	Oval	Dull-brown	2.8	All of US
Polygonum hydropiper	Marshpepper smartweed	Ovoid	Dark-brown, black	2-3.5	All but southern GA and FL
Polygonum pensylvanicum	Pennsylvania smartweed	Round, flat	Black, shiny	2.2-3.5	Central and eastern US
Polygonum persicaria	Ladysthumb	Round, flat	Black, shiny	2.5-3	All but southwestern US, southern CA to TX
Portulaca oleracea	Common purslane	Oval	Black, whitish scar at one end	0.5	All of US
Potentilla canadensis	Common cinquefoil	Ovoid	NR	1	Eastern US, except southeastern States

Table B-1. Comparison of common weed seed characteristics and distribution in the United States to aldicarb granules.

Scientific Name	Common Name	Shape	Color	Length (mm)	Distribution in United States
Granular aldicarb		Round	Black, shiny	0.4-1.1	NA
Potentilla norvegica	Rough cinquefoil	Ovoid	Light-brown	0.8-1.3	All but southeastern and southwestern US
Potentilla recta	Sulfur cinquefoil	Round, oval	Dark-brown	0.9	Northeastern quarter of US
Prosopis juliflora	Mesquite	Oval	Brown	2.5-7	South central and part of southwestern US
Prunella vulgaris	Healall	Obovate	Brown	1.5	All but north central US
Pureraria lobabta	Kudzu	Ovoid	NR	4	Southeastern US
Ranunculus acris	Tall buttercup	Obovate	Dark-brown	2-3	All but north central US
Ranunculus repens	Creeping buttercup	Obovate	Brownish	2.5-3.5	Northern half of US, except north central States, and distinct area in south central US
Rhus glabra	Smooth sumac	Kidney shaped	NR	4	All but northern parts of MT and ND
Rhus radicans	Poison ivy	Ovoid	NR	3.6	All of US
Rhus toxicodendron	Poison oak	Kidney shaped	NR	4	Southwest and Western coast of US
Richardia scabra	Florida purslane, Mexican clover	Triangular	NR	NR	Southern Atlantic, and Gulf Coast States
Rosa arkansana	Prairie rose	Triangular	Brown	4	Eastern and western US, not in central and extreme southeast
Rubus allegheniensis	Allegheny blackberry	Oblong	NR	1	Northeastern quarter of US
Rumex acetosella	Red sorrel	Oval	Reddish-brown, golden-brown, shiny	1.5	All of US
Rumex crispus	Curly dock	Ovoid	Brown, shiny	2	All of US
Salsola kali	Russian thistle	Triangular	Yellowish	2	Central and western US, and coastal States
Scirpus acutus	Hardstem bulrush	Ovoid	Black, lustrous	2-2.5	All but extreme southeast US
Scleranthus annuus	Knawel	Ovoid	Straw	1-1.3	Eastern US, western coastal States
Senecio jacobaea	Tansy ragwort	Oblong	NR	0.3	Pacific coast WA to northern CA; Pacific coasts of ME, RI
Setaria faberi	Giant foxtail	Ovoid	Greenish	1.5	All but western, northernmost and southernmost US
Sida spinosa	Prickly sida	Ovoid	Dark reddish-brown	1-2	Eastern half of US, except northern most States

Bayer CropScience
Report 201692

Table B-1.   Comparison of common weed seed characteristics and distribution in the United States to aldicarb granules.

Scientific Name	Common Name	Shape	Color	Length (mm)	Distribution in United States
Granular aldicarb		Round	Black, shiny	0.4-1.1	NA
Silene noctiflora	Nightflowering catchfly	Round, oblong	Gray	0.8-1	Northern half of US, some south central States
Sisymbrium altissimum	Tumble mustard	Oblong	Dark reddish-brown	1.5	All of US
Solanum carolinense	Horsenettle, Carolina nettle	Round, flat	Yellowish	1.5	Eastern half of US, and selected western States
Solanum nigrum	Black nightshade	Obovate	Yellow, dark-brown	1.2-2.3	Eastern half of US
Solidago canadensis	Canada goldenrod	Elongated, hairy	NR	1.1	Northeastern quarter of US
Sonchus arvensis	Perennial sowthistle	Oblong	Dark reddish-brown	2-3.5	Northern half of US, isolated southern areas
Sonchus asper	Spiny sowthistle	Obovate	Orange-brown	2-3	All of US
Sonchus oleraceus	Annual sowthistle	Elongated, hairy	NR	2.3-3	All of US
Sorghum halepense	Johnson grass	Oval	Reddish brown	3	Southern half of US, parts of WA, OR
Spergula arvensis	Corn spurry	Round, flat	Dull-black, whitish spots	1-1.5	Most of eastern US, western coast, distinct central area
Sporobolus poiretii	Smutgrass	Oblong	NR	0.9-1	Southeastern US, coast of OR
Stellaria media	Common chickweed	Round, flat	Reddish-brown	1-1.2	All of US
Striga lutea	Witchweed	Kidney shaped	NR	0.1	NC and SC
Taraxacum officinale	Dandelion	Elongated	Yellow to greenish-brown	3-4	All of US
Thlaspi arvense	Field pennycress	Ovoid	Dark reddish-brown, black	2-2.3	All of US
Urtica dioica	Stinging nettle	Ovoid	Yellow, grayish-tan	1-1.5	All but northwest WA to southern TX, and southern GA and FL
Verbascum blattaria	Moth mullein	Ovoid, grooved	Dark-brown	0.8	All of US

Table B-1.   Comparison of common weed seed characteristics and distribution in the United States to aldicarb granules.

Scientific Name	Common Name	Shape	Color	Length (mm)	Distribution in United States
Granular aldicarb		Round	Black, shiny	0.4-1.1	NA
Verbascum thapsus	Common mullein	Ovoid, grooved	Brown	0.8	All but northeastern MT, and northwestern ND
Verbena bracteata	Prostrate vervain	Oblong	Dark-brown	2.1	All of US, except large portion of Rocky Mountains, northern ME, NY, and MN
Verbena stricta	Hoary vervain	Oblong	Dark, Grayish-brown	2.5-3	All but northern NE and NY, southeastern US, and between western coastal States and central US
Vernonia altissima	Tall ironweed	Oblong	NR	0.3	Eastern half of US, except northern border States
Vernonia baldwinii	Western ironweed	Oblong	NR	3	Central US
Veronica arvensis	Corn speedwell	Ovoid	NR	1	All but central US
Veronica officinalis	Common speedwell	Oval, flat	Lemon-yellow	1	Northeastern quarter of US
Veronica peregrina	Purslane speedwell	Oval, flat	Orange-yellow	0.4	Eastern half of US, north central and northwestern US, distinct area in central west US
Vicia angustifolia	Narrowleaf vetch	Round	Brown	3	All of US
Xanthium pennsylvanicum	Common cocklebur	Oblong	NR	1.3	All but ME, NH, VT, and northern NY

Appendix C – Dermal Exposure of Burrowing Mammals to Aldicarb

The potential for dermal exposure is determined by the properties of the contaminant, the medium of exposure (e.g., soil, water), the permeability of the animals integument, the area of the integument in contact with the contaminated media, and the duration and pattern of exposure (EPA, 1993). These data are often not available, making assessment of dermal exposure difficult. Feathers and fur reduce the likelihood of significant dermal exposure by limiting contaminant contact with the skin (Sample *et al.*, 1997). Several studies have investigated the dermal acute toxicity of aldicarb in mammals. These studies have focused on rabbits and rats.

Carpenter and Smyth (1966) reported percutaneous LD50s of 32 and 50 mg a.i./kg b.w. for male and female rabbits, respectively. Rabbits, in this preliminary study, were exposed to moistened (50% water by weight) Temik 10G for 24 hours and were observed for the following 14 days. In a subsequent study, 15-day dermal applications of 20, 10 and 5 mg a.i./kg b.w. were tested on both sexes. Rabbits were abraded (i.e., skin was scraped) at the application site, and both wet and dry applications were tested. Results showed depressed weight gain at the highest dose (wet application). No mortality resulted from the dermal applications of Temik 10G in this experiment.

Striegel and Carpenter (1963) reported a technical grade percutaneous LD50 of 5.0 mg a.i./kg b.w. for male rabbits. This study involved a 24-hour exposure, and a 14-day observation period. Technical grade aldicarb was applied as a 5% solution in propylene glycol. The difference in results between this and the previous study was attributed to the vehicles (water and propylene glycol, respectively), because propylene glycol is a better skin penetrant (Carpenter and Smyth, 1965a). Carpenter and Smyth (1965a) reported an intermediate percutaneous LD50 of 12.5 mg a.i./kg b.w. when the vehicle was dimethyl phthalate. Carpenter and Smyth (1965a) also reported on a 4-hour contact study in which rabbits were exposed to a 5% solution of technical grade aldicarb in toluene. A percutaneous LD50 of 3.5 mg a.i./kg b.w. was determined.

A 24-hour skin application test on rabbits in which Temik 10G was applied with adhesive tape resulted in an LD50 of 141 mg a.i./kg b.w (Carpenter and Smyth, 1965a). A similar 4-hour study with wet application of Temik 10G showed that a percutaneous LD50 would likely fall between 160 and 320 mg a.i./kg b.w., however, statistical analysis was not performed on the data (Carpenter and Smyth, 1965b). In a parallel study involving rats, 80 mg a.i./kg b.w. consistently resulted in greater than 50% mortality, suggesting that a percutaneous LD50 for rats would be less than 80 mg a.i./kg b.w (Carpenter and Smyth, 1965b).

A dermal contact equation was used to determine exposure of burrowing mammals to aldicarb (Hope, 1995). Deer mouse (*Peromyscus maniculatus*), short-tailed shrew (*Blarina brevicauda*), and meadow vole (*Microtus pennsylvanicus*) were the burrowing mammals selected to assess dermal contact with aldicarb. Deer mouse and meadow vole

were selected because they likely occur in row crops treated with aldicarb. Short-tailed shrews are unlikely to occur in row crops because they generally require a well-developed leaf litter to protect them from temperature and moisture extremes (Pruitt, 1959). However, shrews could occur in orchards treated with aldicarb.

The dermal contact equation used to determine exposure of burrowing mammals to aldicarb was:

$$DD_i = \frac{SA_i \cdot P_c \cdot EC \cdot CF \cdot \rho_s}{BW_i \cdot \Theta_i \cdot \Psi_i}$$

Equation 1

where:

DD_i = applied daily dose to ith receptor from dermal contact (mg/kg/day)
SA_i = surface area of ith receptor (cm^2)
P_c = proportion of total surface area in contact with soil (per day)
EC = environmental concentration of the contaminant in soil (mg/kg)
CF = conversion factor, 1 x 10^{-3} (kg/mg)
ρ_s = bulk soil density (mg/cm^3)
BW_i = body weight of ith receptor (kg)
Θ_i = area use factor for ith receptor
Ψ_i = seasonality factor for ith receptor

Surface area was derived using the equation for mammals presented in EPA (1993). Because exposure to burrowing mammals was being considered, the fraction of total surface area in contact with soil was assumed to be one, a conservative assumption. A default fraction of 0.22 is typically used (Hope, 1995). An environmental concentration of 27 mg/kg in soil was used in the model for deer mouse and meadow vole. This value corresponds to the maximum application rate of 4.95 lbs a.i./A for row crops. Because shrews could occur in orchards, an environmental concentration of 55 mg/kg in soil was used which corresponds to the maximum application rate of 10.05 lbs a.i./A for pecans. A bulk soil density of 1,500 mg/cm^3 was used in the equation. Body weights for burrowing mammals – deer mouse, short-tailed shrew, meadow vole – were collected from EPA (1993). The geometric mean of available values for males and females was used for each species (Table C-1).

Table C-1. **Geometric mean body weights used in the dermal contact model (EPA, 1993).**

Species	Body Weight (kg)
Deer mouse	0.0184
Short-tailed shrew	0.0167
Meadow vole	0.0301

Because aldicarb is a fast-acting pesticide and recovery from exposure occurs shortly after removal of the pesticide, a depuration rate constant was applied to the results of the dermal contact equation to derive a body burden value.

$$BB = DD_i \cdot e^{-k} \qquad\qquad \text{Equation 2}$$

where:

BB = body burden (mg/kg b.w.)
DD_i = applied daily dose to ith receptor from dermal contact (mg/kg/day)
k = body burden depuration rate constant (ln(2)/body burden half-life, d)

In studies from the literature, animals not killed during overexposure returned to normal function in approximately 6 hours (Baron and Merriam, 1988; WHO, 1991). Thus, a body burden half-life of 6 hours (0.25 days) is a highly conservative value and was used in Equation 2.

Using Equations 1 and 2 produced aldicarb body burdens of 11.3 mg/kg/day for deer mice, 24.0 mg/kg/day for short-tailed shrew, and 9.68 mg/kg/day for meadow vole. These values are lower than the most sensitive reported percutaneous LD50s of 32 and 50 mg a.i./kg b.w. for male and female rabbits, respectively, that used a field-relevant carrier, (i.e., water) (Carpenter and Smyth, 1966). Carriers such as propylene glycol, dimethyl phthalate and toluene increase skin penetration and thus toxicity of aldicarb. These carriers were not considered to be field relevant, and thus the toxicity test results from studies using these carriers were not considered in this analysis.

Conservative assumptions and uncertainties in the equations and data used to derive the dermal exposure and effects values likely resulted in a large overestimation of risk:

- The percutaneous LD50 overestimates toxicity because burrowing mammals in the field would likely obtain their exposures over a longer period of time.
- Equation 1 was not designed for burrowing mammals, rather it was intended for the estimation of exposure from contact with surface soils. To assess exposure to burrowing mammals, the P_c value was changed from the default of 0.22 to 1. This was likely an overestimation of exposure for burrowing mammals, which do not spend all of their time with their entire body in contact with soil.
- Equation 1 does not consider the importance of fur in decreasing contact between aldicarb and the skin of burrowing mammals.
- The permeability of the burrowing mammal integument was not considered, nor was the pattern of exposure. It was assumed, for example, that mammals only burrow in treated rows. In reality, mammals will burrow throughout a field, thus varying their overall exposure to aldicarb.
- The exposure analysis assumed that spatial and temporal use of aldicarb-treated fields was 100%, a very conservative assumption.
- Maximum application rates were also assumed for this analysis.

When all the evidence is considered, it is unlikely that burrowing mammals are at risk from dermal exposure to aldicarb.

Appendix D – Review of the Effects Data for Aquatic Biota and Birds Exposed to Aldicarb

A. 1 Aquatic Biota

Fish

Lethal acute toxicity values for freshwater fish exposed to aldicarb range from 96 hr LC50 values of 49 µg/L for stickleback (*Gasterosteus aculeatus*) to 45,000 µg/L for channel catfish (*Ictalurus punctatus*) (McElligott, 1999; Schlenk, 1995). Studies on aldicarb toxicity have included a variety of freshwater fish species including bluegill sunfish (*Lepomis macrochirus*), rainbow trout (*Oncorhynchus mykiss*), fathead minnow (*Pimephales promelas*), channel catfish, and others. The most frequently studied freshwater fish species are bluegill sunfish, rainbow trout, and fathead minnow. The most sensitive chronic endpoint reported in the literature is for the survival of early life stages of fathead minnows exposed to aldicarb (Pickering and Gilliam, 1982). This study reported a NOAEC (no observed adverse effects concentrations) of 78 µg/L and a LOAEC (lowest observed adverse effects concentrations) of 156 µg/L following 30 days exposure of larvae-juvenile fathead minnows to aldicarb.

A sensitive freshwater fish species identified from the literature was bluegill sunfish. Ninety-six hour LC50 values for this species range from 52 to 450 µg/L (Mayer and Ellersieck, 1986; Carter and Graves, 1972). The latter study was considered unacceptable by EPA because it did not use a control group. The next highest 96 hr LC50 value reported for bluegill sunfish was 145 µg/L (Knott and Beliles, 1966). This study provides a more acceptable upper bound for acute toxicity values for bluegill sunfish than does the value of 450 µg/L reported by Carter and Graves (1972). Other sensitive species include rosy barb (*Barbus conchonius*), snake-skinned gourami (*Trichogaster pectoralis*), and roach (*Rutilus rutilus*), with 96 hr LC50 values of 459, 350, and 363 µg/L, respectively. Rainbow trout are less sensitive, with 96 hr LC50 values between 560 and 1,730 µg/L (Schlenk, 1995; Thun, 1990b).

Toxicity values reported for fathead minnow range from a 96 hr LC50 of 1370 µg/L to a 48 hr LC50 value of 8,860 µg/L (Geiger *et al.*, 1990; Moore *et al.*, 1998). The former study was conducted under flow-through conditions and the latter under static conditions. Pickering and Gilliam (1982) reported a 96 hr LC50 value of 861 µg/L for fathead minnows exposed to aldicarb under flow-through conditions. In this study, fathead minnows were observed for seven days post exposure, without further addition of aldicarb. During the first four days post exposure, some mortality was noted, but all fish regained the equilibrium they had lost during the 96 hr exposure period. No mortality was observed during the next three days.

Although the 96 hr LC50 for bluegill is much higher than the 30-day LOAEC in the study by Pickering and Gilliam (1982), this is not necessarily an indication that increased exposure duration increases toxicity. The 30-day study was conducted on embryo to

juvenile life stages, while the acute study was conducted on juveniles only, a later life stage. Thus, differences in acute and chronic toxicity values could be due to greater sensitivity of early life stages of bluegill, as reported for most other chemicals (McKim, 1977; Macek and Sleight, 1977).

Gallo *et al.* (1995) reported 20-fold greater toxicity of aldicarb to guppy (*Poecilia reticulata*) than zebra fish (*Danio rerio*). Ninety-six hour LC50 values reported for the two species were 666 and 10,065 µg/L, respectively (Gallo *et al.*, 1995).

The ability of different fish species to biotransform and eliminate aldicarb from their systems can affect their sensitivity to the substance. Flavin-containing monooxygenases (FMO) have been associated with the biotransformation and toxicity of aldicarb to fish species. The relationship was compared between a fish species without FMO (channel catfish) and one with FMO (rainbow trout) (Schlenk, 1995). FMO is capable of metabolizing aldicarb to aldicarb sulfoxide, a more potent AChE inhibitor (National Institute of Health, 1979). Ninety-six hour LC50 values reported in this study for channel catfish and rainbow trout were 45,000 and 560 µg/L, respectively (Schlenk, 1995). The author suggested that the decreased bioactivation of aldicarb to aldicarb sulfoxide in channel catfish acts as a protective mechanism against acute toxicity (Schlenk, 1995; Perkins and Schlenk, 2000). The existence of FMO in rainbow trout, bluegill, and medaka (*Oryzias latipes*), may explain their increased sensitivity to aldicarb toxicity (Schlenk, 1995). Although aldicarb sulfoxide is a more potent inhibitor of AChE than is the parent compound at the tissue sites of action (i.e., muscle, brain and serum AChE), the former is about 4 to 30 times less toxic to fish than is the parent compound when fish are exposed to the substances in an external medium (i.e., water). For example, the 96 hr LC50 values for rainbow trout is between 560 and 1,730 µg/L for aldicarb (Schlenk, 1995; Thun, 1990b), but is 7440 µg/L for aldicarb sulfoxide (Odin-Feurtet, 1998). This apparent anomaly is likely due to differences in rates of uptake and/or metabolism in the liver prior to reaching the tissue sites of action. However, no studies were found that explicitly addressed this issue.

From their study using channel catfish, Perkins and Schlenk (2000) and El-Alfy *et al.* (2001) concluded that inhibition of muscle AChE was the ultimate cause of mortality of fish from exposure to aldicarb. Loss of muscular control leads to loss of swimming control and the cessation of opercular movement. The inability to move water across the gills limits the oxygenation of blood, and can lead to hypoxia-induced death (Zinkl *et al.*, 1987).

Invertebrates

Values for the acute toxicity of aldicarb to freshwater invertebrates range from a 48 hr LC50 of 20 µg/L for *Chironomus tentans* to a 96 hr LC50 of >320,000 µg/L for Asiatic clam (*Corbicula fluminea*) and mussels (*Elliptio complanata*) (Moore *et al.*, 1998; Moulton *et al.*, 1996).

Daphnia magna has been the most common species studied. Temperature-dependant 48 hr LC50 values for this species range from 75 to 1,450 µg/L (Song *et al.*, 1997). The

former value was observed at a water temperature of 27°C and the latter at 20°C. Vilkas (1977) calculated a 48 hr LC50 value of 411 μg/L for *D. magna* exposed to aldicarb at an average water temperature of 17°C. The highest toxicity value observed in the literature for *D. magna* was a 48 hr EC50 value of 1,450 μg/L (Thun, 1990a). A chronic 21-day EC50 of 90 μg/L for immobilization was reported for *D. magna* exposed to aldicarb (Handley *et al.*, 1991). This study estimated an EC50 for reproduction between 180 and 560 μg/L. No significant difference in number of young produced per female occurred between controls and the test group exposed to 180 μg/L. No young were produced in the test group exposed to 560 μg/L. Thus, the 21-day NOAEC for reproduction was considered to be 180 μg/L; no LOAEC could be derived from this study. Acute toxicity values for *D. laevis* are lower than those for *D. magna*, with 48 hr EC50 values ranging from 51 to 65 μg/L and 48 hr LC50 values ranging from 70 to 209 μg/L (Foran *et al.*, 1985).

Chironomus sp. were the most sensitive freshwater invertebrates identified from the literature. A 48 hr LC50 value of 20 μg/L was reported for *Chironomus tentans* (Moore *et al.*, 1998). This value was approximately two orders of magnitude lower than the equivalent endpoints reported for *D. magna* (583 μg/L) and scud (*Hyalella azteca*) (3990 μg/L) conducted under very similar laboratory conditions (Moore *et al.*, 1998). Exposure of *Chironomus riparius* to aldicarb resulted in a 24 hr IC50 (for inhibition of ChE) and EC50 values of 27 and 23 μg/L, respectively (Sturm and Hansen, 1999). The equivalent values reported for *D. magna* in this study were 228 and 95 μg/L, respectively (Sturm and Hansen, 1999). Twenty-four hour LC50 values ranging from 17 to 28 μg/L were reported for *Chrionomus thummi* (Suorsa and Fisher, 1986). Ninety-six hour LC50 values reported for the pond snail and apple snail were 11,500 and 175,000 μg/L, respectively (Singh and Agarwal, 1981).

The highest 96 hr LC50 values were reported for the Asiatic clam (*Corbicula fluminea*) and mussel (*Elliptio complanata*) with 96 hr LC50 values >320,000 μg/L (Moulton *et al.*, 1996). This study did not observe a difference in toxicity between test groups exposed to aldicarb in test water measuring 21, 24, or 27°C. One of three mussels exposed to aldicarb at 30°C died within 48 hr. The remaining two mussels in this treatment exhibited gaped shells that closed slowly in response to being tapped (Moulton *et al.*, 1996).

Aldicarb Sulfoxide

Aldicarb sulfoxide is less toxic to freshwater fish species than the parent compound, aldicarb. Acute toxicity values reported for bluegill and rainbow trout range from a 72 hr LC50 of 4,000 to a 96 hr LC50 of 7,440 μg/L, respectively (Clarkson and Hensley, 1968; Odin-Fuertet, 1998). Acute toxicity values reported for freshwater invertebrate species exposed to aldicarb sulfoxide are similar to those reported for aldicarb. Values reported for adult *Daphnia laevis* range from a 48 hr EC50 of 43 μg/L to an adult 48 hr LC50 of 103 μg/L (Foran *et al.*, 1985). *Daphnia magna* are less sensitive with 72 hr LC50 values ranging from 696 to 800 μg/L (Handley *et al.*, 1995; MRID 45592117). No acute toxicity data were identified for saltwater fish and invertebrate species exposed to aldicarb sulfoxide. No chronic toxicity data were identified for freshwater aquatic biota.

Aldicarb Sulfone

Acute toxicity values reported for freshwater fish species exposed to aldicarb sulfone are higher than those for freshwater fish exposed to aldicarb. Rainbow trout and bluegill sunfish exposed to aldicarb sulfone were observed to have 96 hr LC50 values of 42,000 and 53,000 μg/L, respectively (Union Carbide Corporation, 1975). Foran *et al.* (1985) reported acute toxicity values for *Daphnia laevis* ranging from a 48 hr EC50 of 369 to a 48 hr LC50 of 1,124 μg/L. Similar toxicity values were reported for *Daphnia magna*, with 48 hr LC50 values of 280 and 550 μg/L (Union Carbide Corporation, 1975; Handley *et al.*, 1994). No acute toxicity data were identified for saltwater fish and invertebrate species exposed to aldicarb sulfone. No chronic toxicity data were identified for freshwater aquatic biota.

A.2   Birds

Lethal acute toxicity values for birds exposed to technical grade aldicarb range from an oral LD50 of 0.75 mg/kg for the house sparrow (*Passer domesticus*) and common grackle (*Quiscalus quiscula*) (Schafer and Brunton, 1979) to a 5-day dietary LC50 of 786 mg/kg for 21-day-old Japanese quails (*Coturnix coturnix japonica*) (Hill and Camardese, 1984). Studies on aldicarb toxicity have included a variety of bird species including mourning dove (*Zenaida macroura*), red-winged blackbird (*Agelaius phoeniceus*), northern bobwhite (*Colinus virginianus*), common pheasant (*Phasianus colchicus*), and others. A search of the literature revealed three general types of studies used to evaluate the hazard of aldicarb to birds: field exposure studies, dietary studies and gavage studies.

Field Studies

The earliest studies on the effects of aldicarb to the survival of northern bobwhite involved environmental field exposures (Clarkson *et al.*, 1969; Clarkson and Rowe, 1970; Haines, 1971a), which involved placing birds in open-bottom pens over fields treated with Temik® (10% a.i.) formulations. Various application techniques and application rates were tested. In the earliest study with northern bobwhite, Clarkson *et al.* (1969) determined that deeper incorporation or irrigation during application prevented mortality. In a later study by Clarkson and Rowe (1970), no mortality occurred in plots that were irrigated whereas mortality ranged from 33 to 100% when irrigation was not applied. These studies were found unacceptable by the EPA because the exposure conditions in these field were highly unrealistic and did not apply to real world conditions. No endpoints were calculated.

Haines (1970a,b,c) conducted similar field studies investigating the hazard of Temik® 10G (10% a.i.) on common pheasant. Sugar beat fields were treated via in-furrow, banded, and side dressing application methods, at time of planting and during the post-emergence period, both with and without irrigation. All trials resulted in no bird mortalities. In a separate study (Haines, 1971b) immature pheasants appeared to be more susceptible than adults to field treatment, as no treatment related mortalities were observed for adult pheasants. In comparable studies, valley quail (*Lophortyx californicus*) appeared to be slightly more sensitive to aldicarb than common pheasant (Haines, 1970a,b,c, 1971b).

Dietary Studies

Dietary studies primarily involve incorporation of technical grade aldicarb in the feed of pen-reared birds. Dietary studies on aldicarb toxicity have been performed using a variety of bird species including the mallard duck (*Anas platyrhynchos*), common pheasant (*Phasianus colchicus*), and Japanese quail (*Coturnix coturnix*), with LC50 values ranging from >300 to <1000 mg/kg.

Hill *et al.* (1975) tested both ten and five-day-old pen-reared mallards. In a limit test it was determined that the LC50 for the ten-day-old birds was <1000 mg/kg. A 5-day dietary LC50 of 594 mg/kg was calculated for the five-day-old mallards. Hill *et al.* (1975) also studied the dietary toxicity of technical grade aldicarb to the common pheasant (10-day-old) and Japanese quail (14-day-old). For the common pheasant, no mortality occurred at the highest concentration tested and it was concluded that the dietary LC50 must be >300 mg/kg. A dietary LC50 of 381 mg/kg was determined for the Japanese quail. Hill and Camardese (1981) reported similar LC50s of 355 mg/kg and 542 mg/kg for 7 and 14-day-old Japanese quail, respectively. In this test, fourteen and 21-one-day old birds were found to have significantly higher LC50s than those of one and seven-day-old birds.

Gavage Studies

Gavage studies have been more variable, involving both technical grade (95-99.9% a.i.) and Temik® formulations of aldicarb, and both pen-reared and wild birds. Kendall (1990) investigated the acute oral toxicity of technical grade aldicarb to wild mourning doves. Results of this range-finding test indicated that the mourning dove is among the most sensitive species of birds for which aldicarb toxicity data are available with an LD50 value of 0.805 mg/kg. Kendall (1990) conducted two additional studies of the mourning dove, in which birds were released for observation after being dosed. Results of the two experiments were used to establish an LD50 of 1.0 mg/kg. Hawkes *et al.* (1996) re-evaluated the results of the last two Kendall (1990) experiments relating to released birds. An LD50 of 0.82 mg/kg was calculated, slightly lower than the final LD50 reported by Kendall (1990).

Another sensitive bird species identified from the literature was the mallard duck. In a study of adult mallards exposed to technical grade aldicarb, the 24-hour and 14-day LD50s were equivalent, and were determined to be 1 mg/kg b.w (Beavers and Fink, 1979). A wide range of acute oral LD50 values for technical grade aldicarb have been reported for various other bird species; they are 0.75 mg/kg b.w. for the common grackle, 3.16 mg/kg b.w. for the common pigeon, 4.21 mg/kg b.w. for the starling and Japanese quail (Schafer and Brunton, 1979), and 9.5 mg/kg b.w. for the white leghorn cockerels (*Gallus gallus*) (West and Carpenter, 1965).

Hill and Camardese (1984) investigated toxicity to caged adult northern bobwhites in a comparison of technical grade versus granular formulations. The gavage study was used to establish an LC50 of 2.5 mg a.i./kg b.w. for granular Temik® 15G (15% a.i.) and an LD50 of 2.0 mg a.i./kg b.w. for technical grade aldicarb. In a later study, Hawkes *et al.*

(1996) calculated an LD50 of 1.48 mg/kg b.w. for the northern bobwhite exposed to technical grade aldicarb, which is lower than all other reported oral LD50s for the species, but within the 95% confidence interval for technical grade aldicarb reported by Hill and Camardese (1984).

Sterner and Chibanguza tested analytical grade aldicarb on Japanese quail. The LD50 was 4.37 mg/kg b.w. for female birds and 7.29 mg/kg b.w. for male birds with a combined LD50 of 5.59 mg/kg b.w. Ross et al. (1977) observed similar results with three types of 10% a.i. aldicarb granule formulations, with LD50 values of 6.68, 3.76, and 4.04 mg a.i./kg b.w. for the gypsum, corncob, and coal formulations, respectively.

Other comparisons of technical grade versus granular formulations have been found in the literature. In general, LD50 values are lower for technical grade aldicarb than for aldicarb formulations. Grolleau (1999) studied the oral toxicity of technical grade aldicarb and Temik[®] 10G in wild house sparrows. The LD50 of 1.5 mg/kg calculated for Temik[®] 10G (10% a.i.) appeared comparable to technical grade aldicarb although no calculations were provided. For the house sparrow, Balcomb et al. (1984) calculated a granular LD50 of 3.8 mg a.i./kg b.w. using Temik[®] 15G (15% a.i.), higher than and LD50 of 0.75 mg/kg b.w. calculated by Schafer and Brunton (1979) for technical grade aldicarb. Medd et al. (1972) established a 24-hour single oral dose LD50 of 15.2 mg a.i./kg for adult pheasant exposed to a 7.5% a.i. Temik[®] formulation, higher than an LD50 of 5.34 mg/kg for female adult birds exposed to technical grade aldicarb (Hudson et al., 1984).

Aldicarb Sulfone
Acute toxicity values reported for birds exposed to aldicarb sulfone are higher than those for birds exposed to parent aldicarb. Fink (1976a) conducted a dietary study to establish an aldicarb sulfone LC50 for 14-day-old northern bobwhites. A five-day dietary LC50 of 5706 mg/kg was reported. In a similar study, Gallagher et al. (1998b) established a dietary LC50 of 3685 mg/kg for the northern bobwhite. Fink (1976b) conducted a comparable dietary study involving 14-day-old mallards. The 5-day LC50 was reported as >10,000 mg/kg. These results indicate that mallards are more tolerant of aldicarb sulfone than are bobwhites.

Aldicarb Sulfoxide
Aldicarb sulfoxide is less toxic to birds than the parent compound, aldicarb. Gallagher et al. (1998a) conducted a dietary study during which northern bobwhite were exposed to aldicarb sulfoxide during a 5-day period, with a resulting LC50 of 362 mg/kg. For this endpoint, aldicarb sulfoxide was an order of magnitude more toxic than aldicarb sulfone in the feed of fourteen-day-old northern bobwhites. Unfortunately, no comparable studies involving parent aldicarb in diets of bobwhites were available for comparison. Fink (1976c) investigated the oral acute toxicity of aldicarb sulfone to mallards. Fourteen-day-old ducks were treated with one of five doses administered via intubation. The LD50 was 33.5 mg/kg b.w. for aldicarb sulfone, one order of magnitude greater than those reported for parent aldicarb in comparable gavage studies with mallards.

Appendix E – Evaluation of Studies Available from the Open Literature

This appendix presents the data screening criteria that were used to ensure that only high quality data were used in the effects determinations. The criteria distinguish studies as primary, secondary, or unacceptable, according to whether or not sufficient information was provided to evaluate the study design, execution, analyses, results, and other key aspects of the study. Studies deemed acceptable by EPA for use in EFED assessment (EPA, 2006) were accepted for use in the effects determinations, unless erroneous values or other factors raised concerns about data quality.

Data Screening Criteria

Author(s): Handley, J.W., I.G. Sewell and A.J. Bartlett

Title: An assessment of the effect of aldicarb on the reproduction of *Daphnia magna*

Source: Safepharm Laboratories

Compound: Aldicarb

Species tested: *Daphnia magna*

Test duration: 21 days

Endpoint: Reproduction

Based on information available for the data being evaluated, respond to the following criteria using Yes (Y), No (N), Not Reported (NR) answers.

Generic Criteria

Y	Laboratory study with single contaminant exposure only.
Y	Ecologically significant endpoints (survival, growth, or reproduction).
Y	Exposure route through water or sediment (not injection).
NR	Concentration-dependent response demonstrated.
NR	Adequate statistical design to estimate toxic effect concentrations.
Y	Study employed acceptable laboratory practices.
NR	Relevant environmental variables measured and reported.
NR	Survival of controls measured, reported, and acceptable.

Primary Studies
Meets generic criteria, plus:

N	Flow-through test, and
Y	Measured pesticide concentrations.

Secondary Studies
Meets generic criteria, but:

Y	Static test, or
NA	Estimated pesticide concentrations.

Unacceptable Studies

	Studies that do not meet the generic criteria.

Overall study quality:

Primary ___   Secondary _Y__   Unacceptable ___

Studies that do not meet the generic criteria are classified as unacceptable and will not be used in the assessment.

Comments:

A summary of this study was available. The summary states that the study was conducted according to OECD guideline No. 202, that there were no significant differences in number of young produced per adult between the controls and concentrations up to 0.18 mg/L. The study was performed under semi-static conditions. The mean percentage of measured test concentrations was 106% of nominal. Although the observed NOEC was 0.18 mg/L for reproduction, the authors reported a prolonged toxicity NOEC for adults of 0.018 mg/L. Inspite of the lack of information for some of the generic criteria, this study was deemed acceptable for use in the screening-level effects determination.

Reviewer (initials): _RT__

Data Screening Criteria

Author(s): Moore *et al.*, 1998

Title: Comparative toxicity of chlordane, chlorpyrifos, and aldicarb to four aquatic testing organisms.

Source: Archives of Environmental Contamination and Toxicology 34:152-157

Compound: Aldicarb

Species tested: Fathead minnow (*Pimephales promelas*), Scud (*Hyallela azteca*), *Daphnia magna*

Test duration: 96 hr: Fathead minnow

48 hr: Scud, *Daphnia magna*

Endpoint: Survival

Based on information available for the data being evaluated, respond to the following criteria using Yes (Y), No (N), Not Reported (NR) answers.

Generic Criteria

Y	Laboratory study with single contaminant exposure only.
Y	Ecologically significant endpoints (survival, growth, or reproduction).
Y	Exposure route through water or sediment (not injection).
Y	Concentration-dependent response demonstrated.
Y	Adequate statistical design to estimate toxic effect concentrations.
Y	Study employed acceptable laboratory practices.
N	Relevant environmental variables measured and reported.
N	Survival of controls measured, reported, and acceptable.

Primary Studies
Meets generic criteria, plus:

N	Flow-through test, and
Y	Measured pesticide concentrations.

Secondary Studies
Meets generic criteria, but:

Y	Static test, or
N	Estimated pesticide concentrations.

Unacceptable Studies

Y	Studies that do not meet the generic criteria.

Overall study quality:

Primary ____   Secondary ____   Unacceptable _Y_

Studies that do not meet the generic criteria are classified as unacceptable and will not be used in the assessment.

Comments:

Reviewer (initials): __RT__

Data Screening Criteria

Author(s): Pickering and Gilliam, 1982

Title: Toxicity of aldicarb and fonofos to the early-life-stage of the fathead minnow

Source: Archives of Environmental Contamination and Toxicology 11:699-702

Compound: Aldicarb

Species tested: Fathead minnow (*Pimephales promelas*)

Test duration: 96 hr

Endpoint: Survival

Based on information available for the data being evaluated, respond to the following criteria using Yes (Y), No (N), Not Reported (NR) answers.

Generic Criteria

Y	Laboratory study with single contaminant exposure only.
Y	Ecologically significant endpoints (survival, growth, or reproduction).
Y	Exposure route through water or sediment (not injection).
Y	Concentration-dependent response demonstrated.
Y	Adequate statistical design to estimate toxic effect concentrations.
Y	Study employed acceptable laboratory practices.
Y	Relevant environmental variables measured and reported.
Y	Survival of controls measured, reported, and acceptable.

Primary Studies
Meets generic criteria, plus:

Y	Flow-through test, and
Y	Measured pesticide concentrations.

Secondary Studies
Meets generic criteria, but:

| N | Static test, or |
| N | Estimated pesticide concentrations. |

Unacceptable Studies

| | Studies that do not meet the generic criteria. |

Overall study quality:

Primary _Y__   Secondary ___   Unacceptable __

Studies that do not meet the generic criteria are classified as unacceptable and will not be used in the assessment.

Comments:

Reviewer (initials): __RT__

Appendix F – Fischer and Bowers (1997) Data Used to Derive the Nomogram for Calculating Granular Pesticide Residues on Terrestrial Invertebrates

Table F-1. Invertebrate residue levels (wet wt. in mg/kg) resulting from soil incorporated applications.

Site	Appl. Rate (lbs ai/A)	Residue (mg/kg)	Nomogram Values* (lbs ai/A)	Crop	Collection Method	Compound	Study No.	Application Method
1	15.40	0.3	0.02	potatoes	pitfall	fensulfothion	1	band
2	15.40	0.2	0.01	potatoes	pitfall	fensulfothion	1	band
3	7	2.4	0.34	sweet potatoes	pitfall	fensulfothion	2	band
4	7	1	0.14	sweet potatoes	pitfall	fensulfothion	2	band
5	15.40	0.1	0.01	potatoes	pitfall	disulfoton	3	in-furrow
6	15.40	0.2	0.01	potatoes	pitfall	disulfoton	3	in-furrow
7	15.40	0.1	0.01	potatoes	pitfall	disulfoton	3	in-furrow
8	15.40	0.2	0.01	potatoes	pitfall	disulfoton	3	in-furrow
9	15.40	0.4	0.03	potatoes	pitfall	disulfoton	3	in-furrow
10	15.40	5.2	0.34	potatoes	pitfall	disulfoton	3	in-furrow
11	15.40	0.5	0.03	potatoes	pitfall	disulfoton	3	in-furrow
12	20	1.4	0.07	citrus	pitfall	fenamiphos	9	band
13	20	14	0.70	citrus	pitfall	fenamiphos	9	band
14	20	9.7	0.49	citrus	pitfall	fenamiphos	9	band
15	0.643	0.1	0.16	corn	pitfall	tebupirimphos	14	band
16	0.643	0.1	0.17	corn	pitfall	tebupirimphos	14	band
17	0.643	0.1	0.22	corn	pitfall	tebupirimphos	14	band
18	0.643	0.1	0.16	corn	pitfall	tebupirimphos	14	band
19	0.643	0.1	0.16	corn	pitfall	tebupirimphos	14	band
20	0.643	0.1	0.16	corn	pitfall	tebupirimphos	14	band
21	0.643	0.1	0.16	corn	pitfall	tebupirimphos	14	band
22	0.643	0.8	1.28	corn	pitfall	tebupirimphos	14	band

Mean 0.211
Standard Deviation 0.286
Geometric 0.06982

Table F-1. Invertebrate residue levels (wet wt. in mg/kg) resulting from soil incorporated applications.

Site	Appl. Rate (lbs ai/A)	Residue (mg/kg)	Nomogram Values* (lbs ai/A)	Crop	Collection Method	Compound	Study No.	Application Method
		Mean						
		Median	0.156					
		Max	1.28					
		Min	0.00649					

Note: Data are from Bayer unpublished reports.
*Nomogram values are normalized to mg/kg per 1 lb ai/A by dividing the pesticide residues (mg/kg) measured on terrestrial invertebrates in the treated field by the application rate (lbs ai/A).

Appendix G – Weight of Juvenile (≤ 6.5 cm) and Adult (≥ 10 cm) California red-legged frogs from a ten-year (1991-2000) study of four streams in San Luis Obispo County, California (Scott and Rathbun, 2001).

Table G-1. Weight of juvenile and adult California red-legged frogs used in *TDI* Model.

Juvenile (g ww)						Adult (g ww)						
4.3	9.2	94.5	83.5	125	114.8	114	120	147	121	155	152.5	200
4.5	11.5	89	93	80	112	128	118	118	153	183	136	209
4.5	11.0	101	110	95.5	102	113	95	129	152	190	116	153
4.5	9.0	97	94	114	121	78	104	128	167	134.5	160	154
6.0	11.0	91	89	99	101	99	128	117	127	150	172	158
6.0	11.7	90	102	105	108.5	111	112	125	172	151	200	235
7.5	10.5	99.5	118	76	107	135	129	133	184	116	180	182
7.0	11.7	94.5	108	98	98.5	129	159	148	203	148	226	191
7.0	14.5	116	92.5	102	89	113	122	142	143	176	140	188
7.7	12.3	101	101	91	101	102	115	142	152	164	168.5	183
7.5	14.0	99	94	112	126	118	128	130	151	140	162	210
6.5	13.5	82.5	92	118	78.5	118	143	126	150	137	157	205
9.3	13.5	84	115	99	119	113.5	118	123	150	148	144.5	182
7.4	18.0	96	89	98	125	98	142	144	150	167	126	183.5
7.2	16.0	105	123	74	98	135	118	184	162	178	180	219
8.6	14.5	112	137	130	86	101	121	121	139	160	139	194
8.2	14.0	96	113	104.5	103.5	102	114	131	139	179	165	168
8.5	15.0	96	122	121	108	132	117	132	153	179	152	235
6.8	17.5	93	114	94.5	113	120	124	110	175	120	175	165
8.5	15.5	98	96	101.5	107	105	123	127	152	132	177	207.5
7.8	17.5	89	86	107	96	126	107	137	112	178	165	208
8.5	21.0	80	104	107	121	114	123	160	160	195	185	187.5
7.8	21.0	93	94	101	90.5	139	129	123	136	167	171	162.5
9.5	16.0	98	126	113	110	97	125	158	158	153	202	197

Bayer CropScience
Report 201692

6.1	17.0	81.5	96	134	113	122	115	107	133	157	200	247
8.4	19.0	123	104	89.5	118	119	138	121	177	162	194	223
8.6	21.5	90	95.5	97	111	122	120	129	119	168	142	210.5
10.9	20.0	100	105	109	123	128	140	159	167	179	197	
8.3	21.5	89	102	109	125	117	117.5	130	153	175	155	
9.5	20.5	108	96	94	122	115	121	109	191.5	145	178	
8.6	23.5	102	91	98.5	122	123	114	158	150	181	212	
11.0	19.0	100	94.5	111	99	123	118	128	160	124	168	
8.6	24.5	92	107.5	87.5	122	124	137	140	176	174	202	
8.0	27.0	106	103	102	111	119	134	159	120	154	186	
10.5	22.0	98	103	119	104	135	124	131	167.5	170	160	
10.5	23.5	91.5	86	96	94	143	170	147	166	195	179	
Mean = 12.2g												

Mean = 131.9g